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Macroinvertebrados bentônicos como ferramenta na avaliação da qualidade ecológica de reservatórios tropicais

Joseline Molozzi

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MACROINVERTEBRADOS BENTÔNICOS COMO FERRAMENTA NA AVALIAÇÃO DA QUALIDADE ECOLÓGICA DE RESERVATÓRIOS TROPICAIS

Tese apresentada ao Programa de Pós-graduação em Ecologia Conservação e Manejo de Vida Silvestre como requisito parcial para a obtenção do título de Doutor.

Joseline Molozzi

Orientador:
Prof. Dr. Marcos Callisto
(Departamento de Biologia Geral, ICB, UFMG)

Co-orientador:
Prof. Dr. João Carlos Marques
(Departamento de Zoologia-IMAR, Coimbra, Portugal)

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Dedico essa tese às pessoas que verdadeiramente

me ajudaram na realização deste sonho:

em especial aos meus pais.

"É melhor tentar e falhar que preocupar-se e ver a vida passar,
é melhor tentar, ainda que em vão, que sentar-se fazendo nada até o final.
Eu prefiro na chuva caminhar, que em dias tristes em casa me esconder.
Prefiro ser feliz, embora louco, que em conformidade viver....."
Martin Luther King

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RESUMO

A avaliação da qualidade ecológica de ecossistemas aquáticos tem sido importante para gestores e pesquisadores. Assim, o objetivo geral desta tese foi adaptar e desenvolver diferentes metodologias com o intuito de gerar ferramentas para avaliar a qualidade ecológica de reservatórios urbanos, utilizando comunidade de macroinvertebrados bentônicos na região litorânea. Para isso: i) foi verificado se deformidades morfológicas no aparelho bucal podem ser utilizadas como uma ferramenta em avaliação ambiental; ii) foi estabelecido através de relações alométricas um modelo estatístico para estimar biomassa da espécie exótica *Melanoides tuberculatus* (Thiaridae, Gastropoda); iii) foi avaliada a influência da complexidade estrutural de habitats sobre as comunidades de macroinvertebrados bentônicos; iv) foi proposta uma abordagem de seleção de locais com máximo potencial ecológico em reservatórios urbanos; v) foi desenvolvido, testado e proposto a avaliação de qualidade de água com base em um modelo estatístico preditivo, bem como proposta a avaliação de qualidade de água em classes de estado ecológico; e vi) foi comparado e testado o desempenho de índices de diversidade e índices de eco-exergia e eco-exergia específica para reservatórios. Esta tese é composta por 6 capítulos com os dados amostrados ao longo de dois anos (2009 - 2010) em três reservatórios urbanos na bacia hidrográfica do rio Paraopeba, Minas Gerais. O Capítulo 1 refere-se à avaliação de deformidades morfológicas no aparelho bucal *mento* de larvas de Chironomidae (Diptera, Insecta), em três reservatórios sob diferentes impactos antrópicos. O maior percentual de deformidades no *mento* das larvas foi encontrado no reservatório de Serra Azul na estação seca (6,9%), enquanto que nos reservatórios de Ibirité e Vargem das Flores as porcentagens de deformidades foram menores que 6%. A elevada porcentagem de deformidades encontrada em indivíduos da subfamília Chironominae em Serra Azul (>6%) pode ser uma resposta à alta concentração de manganês presente no sedimento do reservatório, devido à natureza geológica da rocha matriz. Assim, concluímos que as deformidades no aparelho bucal de larvas de Chironomidae não podem ser utilizadas como um indicador de impacto, pois estas deformidades podem estar associadas a origens naturais (fatores genéticos, ou mesmo em função da composição química do sedimento). No Capítulo 2 foi proposto um modelo estatístico para estimar a biomassa dos *Melanoides tuberculatus*. Foram obtidas as medidas de comprimento total e abertura da concha de 70 indivíduos. Estas medidas foram correlacionadas com os valores de biomassa para construção dos modelos: exponencial e potencial, ambos apresentaram elevados coeficientes de determinação. O modelo exponencial foi o melhor preditor de biomassa, apresentando um coeficiente de determinação acima de 93%. A modelagem utilizada neste estudo oferece uma ferramenta importante para determinar a biomassa de *M. tuberculatus* em reservatórios eutróficos brasileiros. O Capítulo 3 refere-se à avaliação da influência da complexidade estrutural dos habitats sobre as comunidades de macroinvertebrados bentônicos. Para isso foi adaptado e modificado o protocolo de diversidade de habitats físicos proposto pela USEPA. Os resultados das variáveis mensuradas pelo protocolo de diversidade de habitats evidenciaram que o reservatório de Serra Azul sofre influência significativa do sub-bosque (decidual-arbustivo) e do ângulo de inclinação do barranco. Nos reservatórios de Ibirité e Vargem das Flores as variáveis que apresentaram uma relação significativa com as comunidades de macroinvertebrados bentônicos foram: cobertura do solo, influência humana e presença de macrófitas aquáticas. As informações obtidas com a utilização do protocolo de diversidade de habitats é, portanto, uma ferramenta complementar no monitoramento de reservatórios, ajudando a orientar programas de restauração de características físicas e químicas de reservatórios urbanos. No Capítulo 4 foi realizada a seleção de locais de referência (máximo potencial ecológico) com base na classificação *a posteriori* utilizando as variáveis de pressão para caracterização de locais de referência. Foram selecionados 28 locais com máximo potencial ecológico, divididos em dois sub-grupos. Os resultados evidenciaram que os indicadores ambientais que melhor explicaram a distribuição das comunidades bentônicas nos locais com máximo potencial ecológico foram: profundidade, cascalho/pedregulho, areia grossa, silte/argila ou lama, do reservatório e silte/argila/lama na região litorânea, sendo que estas variáveis descriminaram 96% da distribuição das comunidades. Estes dois sub-grupos de comunidades biológicas e as respectivas condições ambientais são a base para o desenvolvimento de um futuro sistema de avaliação da qualidade de água em reservatórios tropicais. No capítulo 5 foi testada uma ferramenta de avaliação com base na estrutura de comunidades de macroinvertebrados bentônicos para avaliar o potencial ecológico em reservatórios tropicais. Para isso, foi utilizado um sistema de avaliação conceitual baseado na condição de referência de aproximação e desenvolvido um modelo estatístico baseado em 28 sítios classificados com máximo potencial ecológico. Sessenta e dois locais impactados foram utilizados para testar o modelo. Os resultados mostraram que, com a utilização de 3 classes de classificação do estado ecológico, é possível destinguir de forma significativa os diferentes níveis de pressão a que os locais estão submetidos. Foi demonstrado que as comunidades bentônicas em reservatórios também podem ser utilizadas como elemento de avaliação de qualidade de água. No Capítulo 6 foi testada a capacidade dos índices Eco-exergia, Eco-exergia específica, diversidade de Shannon-Wiener e Margalef em distinguir locais de referência de locais impactados. Os resultados mostraram que os índices de Shannon-Wiener, Margalef e Eco-exergia, foram capazes de distinguir de forma significativa os locais com máximo potencial ecológico dos locais impactados. Assim, observamos que os índices de diversidade e os índices de Eco-exergia são indicadores ecológicos e oferecem informações sobre o estado e a saúde de reservatórios urbanos. Os resultados desta tese evidenciam a necessidade de utilizar indicadores biológicos para avaliar a qualidade de água de reservatórios, e demonstram que também podemos utilizar outras ferramentas, índices e modelos preditivos, para avaliação da qualidade ecológica de reservatórios em programas de biomonitoramento.

Palavras-chave: Macroinvertebrados bentônicos, qualidade ecológica, reservatórios, indicadores biológicos.

ABSTRACT

Assessment of aquatic ecosystem quality is becoming increasingly important for managers and researchers. The general objective of this thesis was to adapt and develop different methodologies aiming toward the development of tools to assess the ecological quality of reservoirs, through the use of the littoral benthic macroinvertebrate communities. To do this: i) morphological deformities in Chironomidae mouthparts were tested for their use as tools in environmental assessments; ii) through allometric relationships, a statistical model was established to estimate the biomass of alien species *Melanoides tuberculatus* (Thiaridae, Gastropoda); iii) the influence of physical habitat structural complexity on benthic macroinvertebrate communities was assessed; iv) an approach was proposed to select sites with maximum ecological potential in urban reservoirs; v) an evaluation of the water quality was developed, tested and proposed, based on a predictive statistical model, as well it was proposed an assessment of water quality in ecological status classes; and vi) an evaluation and comparison of the performance of diversity, eco-exergy and specific eco-exergy indices in reservoirs. This thesis consists of 6 chapters based on data collected over two years (2009 - 2010), in three urban reservoirs in the watershed of Paraopeba, Minas Gerais, Brazil. Chapter 1 describes the morphological deformities in the mentum of Chironomidae larval (Diptera, Insecta) in three reservoirs under different trophic impacts. In Serra Azul, 6.9% of the mentum were deformed in the dry season, whereas in the Ibirité and Vargem das Flores 6% of the mentum were deformed. The higher percentage of deformities in the Chironominae sub-family individuals in Serra Azul (>6%) may be a response to background high concentrations of sedimentary manganese there due to the matrix rock geological origin. Thus, it was concluded that Chironomidae mouthpart deformities can not be used as impact indicators because they may be related to natural origins (genetic factors or even due to the sediments' chemical composition). In Chapter 2, a statistical model was proposed to estimate the biomass of *Melanoides tuberculatus* (Thiaridae, Gastropoda). Measurements of the total length and shell opening of 70 individuals were obtained and correlated with the biomass values for construction of exponential and power-function models. Both models showed high coefficients of determination but the exponential model was the better biomass predictor, with a coefficient of determination over 93%. The modeling used in this study provides an important tool to determine the biomass of *M. tuberculatus* in eutrophic reservoirs. Chapter 3 focused on the influence of habitat structure complexity on benthic macroinvertebrate communities. To assess this, the USEPA physical habitat protocol was adapted and modified. The results of the variables measured by the protocol found that the macroinvertebrate community in Serra Azul was significantly influenced by the shrub understory and bank angle. In Ibirité and Vargem das Flores the variables of land cover, human disturbance and aquatic macrophytes showed significant relation with the benthic macroinvertebrate communities. Information obtained through the use of the habitat protocol provided complementary information for reservoir monitoring and may help to guide reservoir rehabilitation programs. In Chapter 4, reference sites were selected with maximum ecological potential, based on an *a posteriori* classification of pressure variables for the reference sites characterization. Twenty-eight sites with high ecological status were selected and divided in two sub-groups. The results showed that the environmental indicators that best explained the distribution of benthic communities in sites with high ecological status were: bottom substrate type, presence of gravel/boulders, coarse sand, silt/clay or muck, depth and shoreline substrate zone, and these variables discriminated

96% of the distribution of the communities. These two subsets of biological communities and respective environmental conditions are a basis for future development of a quantitative assessment system to monitor tropical reservoirs. Chapter 5 it was tested the value of an assessment tool based on the structure of benthic invertebrate communities to evaluate the Ecological Potential of urban reservoirs. For that, it was designed a conceptual assessment scheme based on the Reference Condition Approach and developed a statistical model based on 28 sites classified as having maximum ecological potential. Sixty-two disturbed sites were used to test the model. The results showed that the use of 3 classes of ecological status classification is possible to significantly distinguish different levels of pressure to which the sites are submitted. It was demonstrated that the benthic communities in reservoirs can also be used as an assessment of water quality. Chapter 6 focused on testing the ability of Eco-exergy, Specific Eco-exergy, Shannon-Wiener, and Margalef indices to distinguish between reference and impaired sites. The results showed that the Shannon-Wiener, Margalef, and Eco-energy indices significantly distinguished reference sites from highly disturbed sites. Therefore, it was noted that the diversity and Eco-exergy indices are ecological indicators and offer information of the state and environmental health of urban reservoirs. The results of this thesis highlight the value of using biological indicators to assess the ecological quality of reservoirs and demonstrate that other tools, indices and other predictive models can also be used to assess the reservoirs ecological quality as a component of biomonitoring programs.

Keywords: Benthic macroinvertebrates, ecological quality, reservoirs, biological indicators.

INTRODUÇÃO

Os reservatórios são ecossistemas modificados, construídos em resposta às demandas de crescimento econômico. No entanto, são componentes diferenciados na paisagem e representam uma inserção nova do ponto de vista de ecossistemas aquáticos, promovendo consideráveis alterações no regime hidrológico e na distribuição ecológica de organismos em rios e em suas bacias hidrográficas (Tundisi, 2006). A instabilidade do novo ambiente formado, fruto não apenas do impacto do represamento, mas também de perturbações produzidas pela urbanização e atividades agrícolas e industriais, tornam as comunidades aquáticas instáveis e gradativamente mais simples (Fore et al., 1994; Klemm et al., 2003).

Entre os organismos aquáticos, as comunidades de macroinvertebrados bentônicos têm sido cada vez mais estudadas na perspectiva de bioindicadores de qualidade de água (Bonada et al., 2006). O uso de bioindicadores para avaliar a qualidade da água é baseado em respostas dos organismos às variações do meio em que vivem, sejam essas perturbações de origem antrópica ou natural (Barbour et al., 1996; Bonada et al., 2006).

O uso de macroinvertebrados bentônicos em programas de biomonitoramento deve-se às características que estes organismos apresentam, tais como: (i) serem de fácil coleta e identificação; (ii) muitos táxons são sedentários e apresentam ciclos de vida longos, sendo portanto capazes de registrar efeitos acumulativos e alterações de habitats (Barbour et al., 1999); (iii) serem sensíveis a alterações físicas e químicas nos diferentes ecossistemas; (iv) e suas respostas a estas alterações serem detectáveis e mensuráveis (Thompson et al., 2007).

Assim, a abordagem tradicional de avaliação de qualidade de água utilizando apenas fatores físicos e químicos tem sido gradativamente substituída por avaliações que englobam as características biológicas dos ecossistemas (Thompson et al., 2007). A legislação brasileira, através da Lei 9.433/97 que estabelece a Política Nacional de Recursos Hídricos, e a Resolução do CONAMA 357/05 artigo 8º parágrafo 3º, prevêm que a qualidade de água poderá ser avaliada por indicadores biológicos. O Estado de Minas Gerais, pelo *Plano Diretor de Recursos Hídricos* desde 1997, e pela Resolução do Conselho Estadual de Política Ambiental (COPAM - 001/2008), vem procurando estabelecer bases de processos de avaliação de suas bacias hidrográficas por meio de indicadores biológicos. A utilização de bioindicadores culmina, deste modo, em uma ferramenta que possibilita uma avaliação da qualidade ecológica dos ecossistemas aquáticos, que poderá ser realizada através da utilização de índices bióticos (Barbour et al., 1999; Bonada et al., 2006).

De acordo com a Abordagem da Condição de Referência (RCA) (Reynoldson et al., 1997; Reynoldson et al., 2001; Bailey et al., 2004; Stoddard et al., 2006), que é utilizada como base de avaliação da Diretiva Quadro da Água na Europa (Directive 2000/60/CE), a integridade de comunidades encontradas em um local deve ser analisada de acordo com os seus desvios para as comunidades esperadas na ausência de perturbações antrópicas (Nijboer et al., 2004; Ruse, 2010; Hawkins et al., 2010). Assim, é fundamental avaliar como são as comunidades biológicas em um dado ecossistema na ausência de impactos.

No caso dos reservatórios a ausência de impactos é impossível, pois estes são corpos d'água fortemente modificados, onde o ambiente passou de um ecossistema lótico para lêntico, e mudanças significativas na estrutura dos rios e suas bacias hidrográficas ocorreram (Tundisi & Matsumura-Tundisi, 2003). Deste modo, observam-se alterações nas propriedades físicas, químicas ou biológicas, resultado de atividades humanas que direta ou indiretamente afetam a saúde dos ecossistemas (CONAMA- Brasil 001/86).

Neste sentido, a condição de referência que é baseada no conceito de *pristino* definido como um "local intocado", não pode ser usada em reservatórios (Stoddart et al., 2006). O estabelecimento de locais com máximo potencial ecológico poderá ser o primeiro passo no desenvolvimento de ferramentas para avaliar a qualidade ecológica de reservatórios urbanos.

No entanto, a utilização de todo um reservatório como referência não é consensual. Navarro et al. (2009) consideram que devido à dificuldade em encontrar reservatórios não poluídos, a utilização de um reservatório que apresente uma boa qualidade ecológica pode ser utilizada como referência para outros reservatórios. Alguns autores defendem que os reservatórios apresentam um gradiente de influência humana, que será refletido em diferentes níveis de impactos (Gibson et al., 2000; Dodds et al., 2006). Portanto, em um mesmo reservatório podem ser diagnosticados locais impactados e outros não, dependendo das atividades no seu entorno, refletindo em um gradiente de impactos ecológicos.

Muitos estudos realizados em reservatórios espanhois (Prat et al., 1991) e em reservatórios brasileiros (Moretto et al., 2003; Moreno & Callisto, 2006) observaram que a riqueza de macroinvertebrados bentônicos em áreas profundas, em geral é muito pobre. Assim, na região litorânea dos reservatórios a riqueza bentônica é mais elevada, devido a maior disponibilidade de habitas físicos, a um substrato mais mais heterogêneo e as concentrações de oxigênio dissolvido são mais elevadas (Thompson et al., 2007).

A União Européia, através da Diretiva Quadro da Água, desenvolveu o conceito de "Estado Ecológico", que sugere à qualidade de comunidades biológicas, de características hidrológicas e de características físicas e químicas da água, e expressa a qualidade estrutural e funcional dos ecossistemas aquáticos (Directive-2000/60/EC) (Fig. 1).

Para avaliar o "Estado Ecológico" de ecossistemas aquáticos podem ser utilizados diferentes índices ecológicos. Estes baseiam-se na idéia de que os organismos exibem variados graus de seletividade de habitats e níveis diferentes de tolerância à poluição (Karr, 1991; Barbour et al., 1996; Marques et al., 2009).

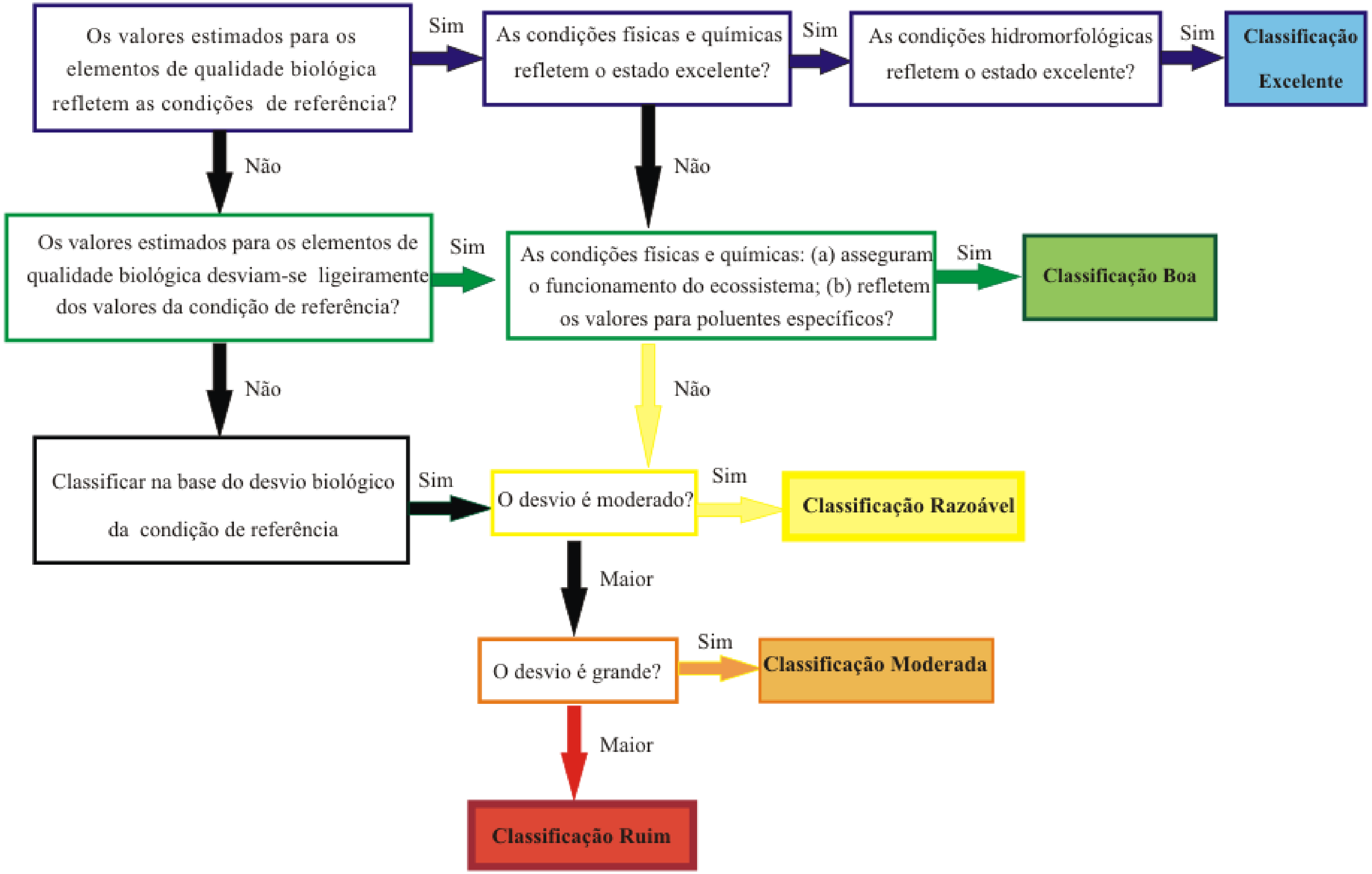

Fig.1 Indicação do papel relativo dos elementos de qualidade de água: biológicos, hidromorfológicos e físicos e químicos na classificação do estado ecológico, de acordo com as definições normativas da Diretiva Quadro da Água (adaptado de Vincent et al., 2003).

Na literatura internacional há inúmeros índices de acordo com suas especificidades, dentre estes, destacam-se: (i) os índices baseados em *espécies indicadoras* que baseiam-se na presença ou ausência de espécies indicadoras, com espécies extremamente sensíveis ao estresse ambiental, tolerando apenas uma faixa estreita de condições (Borja et al., 2000; Smith et al., 2001; Diaz et al., 2004). Nesta categoria, há também os índices com base em espécies bioacumuladoras, descritas como capazes de resistir e acumular diversas substâncias poluentes nos seus tecidos (Storelli & Marcotrigiano, 2001); (ii) os índices baseados em *estratégias ecológicas* que avaliam os efeitos do estresse ambiental sobre as diferentes estratégias ecológicas de organismos aquáticos. É o caso dos índices baseados em grupos funcionais (Belsher, 1982; Dalvin & Ruellet, 2007); (iii) os índices baseados na *diversidade* que consideram a abundância proporcional de espécies, índices de dominância, e índices que levam em conta os aspectos taxonômicos, numéricos, ecológicos, genéticos e filogenéticos de diversidade (Shannon & Weaver, 1963; Margalef, 1968; Clarke & Warwick, 1999; Mackey & Currie, 2001; Magurran, 2004); (iv) os índices baseados na *biomassa e abundância de espécies* baseiam-se na energia contabilizada nos ecossistemas e consideram variações na biomassa e abundância de organismos como uma medida de perturbação ambiental (Warwick, 1986; Warwick & Clarke, 1994); (v) os índices *multimétricos* baseiam-se na tentativa de integrar informações sobre diferentes aspectos do ambiente, incorporando métricas que medem os níveis ecológicos de indivíduos, comunidades, ecossistemas e paisagens. Podem incluir medidas de diversidade, riqueza, composição taxonômica, estrutura trófica do ecossistema, entre outros (Karr, 1991); e (vi) os índices *termodinamicamente baseados em análise de rede* que consideram informações dos ecossistemas em uma perspectiva holística considerando os conceitos de exergia (Jørgensen & Mejer, 1979; Jørgensen & Marques, 2001) e de ascendência (Ulanowicz, 1980).

Os índices ilustram as respostas de comunidades frente a modificações naturais ou antrópicas, e as integram em um único valor que expressa a qualidade ecológica de um ecossistema (Borja et al., 2000; Diaz et al., 2004). O valor numérico sintetiza esta complexidade e pode ser relacionado a uma ampla escala de medidas físicas, químicas, morfológicas e biológicas (Pinto et al., 2009).

Levando em consideração que no Brasil ainda não há uma ferramenta que utilize a comunidade de macroinvertebrados bentônicos na região litorânea para avaliar a qualidade de água em reservatórios, esta tese integra diferentes metodologias para avaliar a qualidade ecológica de reservatórios urbanos, sendo composta por 6 Capítulos. No Capítulo 1 foi avaliado se deformidades morfológicas no aparelho bucal de larvas de Chironomidae são relacionadas a diferentes níveis de impactos a que os reservatórios estão submetidos. No Capítulo 2 foi avaliada a possibilidade de estabelecer a biomassa de *Melanoides tuberculatus* uma espécie exótica, através de um modelo estatístico. No Capítulo 3 foi avaliada a influência da complexidade estrutural de habitats sobre as comunidades de macroinvertebrados bentônicos. No Capítulo 4 foram estabelecidos locais com máximo potencial ecológico como ferramenta para auxiliar em Programas de Biomonitoramento. No Capítulo 5 foram realizadas modificações e adaptações de um modelo preditivo para avaliar o estado ecológico de reservatórios, bem como testadas diferentes classes de qualidade de água. E no Capítulo 6 foram utilizados diferentes índices de diversidade e de eco-exergia para testar a capacidade dos índices em separar locais impactados de locais com máximo potencial ecológico. Desta forma, os resultados obtidos representam uma contribuição para estudos em reservatórios urbanos utilizando comunidades de macroinvertebrados bentônicos na região litorânea, além de fornecer ferramentas que poderão ser utilizadas em Programas de Biomonitoramento no Brasil.

Esta tese é composta pelos seguintes artigos e manuscritos:

1) Morais, S. S., Molozzi, J., Viana, A. L., Viana, T. H. & Callisto, M. 2010. Diversity of larvae of littoral Chironomidae (Diptera: Insecta) and their role as bioindicators in urban reservoirs of different trophic levels. Brazilian Journal of Biology 70(4): 995-1004.

2) Silva, E. C., Molozzi, J. & Callisto, M. 2010. Size-mass relationships of *Melanoides tuberculatus* (Thiaridae, Gastropoda) in an eutrophic reservoir. Revista Brasileira de Zoologia 27(5): 691-695

3) Molozzi, J., França, S. J., Araujo, T. L. A., Viana, T. H., Hughes, R. H. & Callisto (no prelo). Diversidade de habitats físicos e sua relação com macroinvertebrados bentônicos em reservatórios urbanos em Minas Gerais. Iheringia Série Zoologia.

4) Molozzi, J., Feio, M. J., Salas, F., Marques, J. C. & Callisto, M. Maximum ecological potential of tropical reservoirs and benthic macroinvertebrate communities. Submetido à Hydrobiologia.

5) Molozzi, J., Feio, M. J., Salas, F., Marques, J. C. & Callisto, M. Development and test of a statistical model for the ecological assessment of tropical reservoirs based on benthic macroinvertebrates. A submeter à Aquatic Ecology.

6) Molozzi, J., Feio, M. J., Salas, F., Marques, J. C. & Callisto, M. Potential of thermodynamic oriented ecological indicatores as tools for environmental management in tropical reservoirs. A submeter à Ecological Modelling.

OBJETIVOS, HIPÓTESE E PERGUNTAS

Objetivo Geral

Adaptar e desenvolver diferentes metodologias de biomonitoramento com o intuito de obter ferramentas para avaliar a qualidade ecológica de reservatórios utilizando comunidades de macroinvertebrados bentônicos na região litorânea.

Objetivos Específicos

1) Avaliar se deformidades morfológicas em larvas de Chironomidae são relacionadas com diferentes níveis de impacto a que reservatórios urbanos estão submetidos.

2) Estabelecer, através de relações alométricas, uma função matemática para estimar a biomassa de populações de *Melanoides tuberculatus*.

3) Avaliar a influência da complexidade estrutural de habitats sobre comunidades de macroinvertebrados bentônicos.

4) Selecionar locais com máximo potencial ecológico com base em dados de pressão (hidromorfológicos, medidas físicas e químicas da água) e características de comunidades de macroinvertebrados bentônicos.

5) Testar, propor e avaliar a qualidade de água em reservatórios com base em um modelo estatístico preditivo, bem como propor avaliação da qualidade de água em classes de estado ecológico

6) Testar e avaliar a capacidade dos índices de Eco-exergia, Eco-exergia específica, Shannon-Wiener e Margalef em distinguir locais com máximo potencial ecológico e locais impactados em reservatórios urbanos.

Hipótese

O gradiente de impacto a que os reservatórios estão submetidos infuencia a existência de um gradiente na distribuição das comunidades de macroinvertebrados bentônicos, os quais serão utilizados como ferramenta na avaliação da qualidade ecológica de reservatórios.

Perguntas

1) As deformidades morfológicas no *mento* das larvas de Chironomidae estão relacionadas com os diferentes níveis de impacto a que os reservatórios estão submetidos?

2) Através de um modelo estatístico é possível estimar a biomassa de *Melanoides tuberculatus*?

3) As variáveis do protocolo de diversidade de habitat poderão auxiliar no entendimento dos fatores que influenciam a distribuição de macroinvertebrados bentônicos em reservatórios urbanos?

4) Um reservatório localizado em uma área de proteção especial, poderá ser considerado como reservatório de referência?

5) Um modelo estatístico preditivo deverá ser constituído de quantas classes para se avaliar o estado ecológico de reservatórios urbanos?

6) Diferentes indicadores poderão elucidar de forma efetiva locais impactados de locais com máximo potencial ecológico?

ÁREAS DE ESTUDO

Ao todo foram avaliados 90 locais de amostragem na região litorânea de três reservatórios (Ibirité, Vargem das Flores e Serra Azul) localizados na Bacia Hidrográfica do rio Paraopeba, afluente no trecho alto da bacia do rio São Francisco, estado de Minas Gerais (Fig. 2). O clima na região é considerado tropical sub-úmido (Cwb), com chuvas no verão (novembro a abril) e inverno seco (maio a outubro). A temperatura média anual é de ca. 20° C (Moreno & Callisto, 2006).

Os reservatórios de Serra Azul e Vargem das Flores estão inseridos na região do Quadrilátero Ferrífero, circundados pelas Serras do Curral, Cachimbo, Azul e do Itatiaiuçu (Tabela 1).

O reservatório de Ibirité (20°01’13.39” S; 44º06’44.88” W) é formado pela confluência dos rios Pintados e Retiro da Onça. O rio Retiro da Onça apresenta intensa atividade urbana, com contaminação por Coliformes termotolerantes (Pinto-Coelho et al., 2010). O reservatório apresenta acelerado processo de eutrofização artificial, que se caracteriza por acentuada produção primária, com “blooms” de *Microcystis* sp. e de macrófitas *Eichhornia crassipes* (Moreno & Callisto, 2006).

O reservatório de Vargem das Flores (19º54’25.06’’ S; 44º09’17.78” W) é formado pelo represamento do ribeirão Betim. Apresenta ocupação urbana em suas margens e é utilizado para atividades de lazer e pesca. Apresenta algumas regiões com presença de macrófitas aquáticas.

O reservatório de Serra Azul (19º59’24.92” S; 44º20’46.74” W) está inserido na bacia hidrográfica do Ribeirão Serra Azul, limita-se ao Sul com a bacia do Rio Manso.O Ribeirão Serra Azul pertence à bacia do Rio Paraopeba e drena uma área de 263Km^2. Nasce na Serra do Itatiaiuçu e, após confluir com o Ribeirão Mateus Leme, onde passa a receber a denominação de Ribeirão Juatuba, deságua no Rio Paraopeba pela margem esquerda. Tem como principais afluentes pela margem esquerda o Córrego do Tijuco, o Córrego do Brejo, o Córrego Ponte de Palha, o Córrego Ponte d`Areia e o Córrego da Matinha; e pela margem direita, o Córrego do Barreiro, o Ribeirão do Diogo, o Córrego do Gavião, o Córrego Capão do Isidoro, o Córrego Goiabeira, o Córrego do Buracão e o Córrego dos Freitas. A área de proteção da Bacia é de 27.200 ha, sendo que o território de domínio da COPASA (Companhia de Saneamento de Minas Gerais) é de 3.200ha. Possui margens bem preservadas, com vegetação nativa característica do Cerrado, com variações da Mata de Galeria, Cerradão, Campo Sujo, Campo Limpo e Mata Estacional Semidecidual (COPASA, 1980). É considerado, juntamente com o reservatório Vargem das Flores, um dos principais mananciais de abastecimento de água para a região metropolitana de Belo Horizonte pela COPASA-MG (Morais et al., 2010).

Tabela 1. Caracterização dos reservatórios de Serra Azul, Vargem das Flores e Ibirité na bacia hidrográfica do rio Paraopeba, MG.

Características	**Serra Azul**	**Vargem das Flores**	**Ibirité**
Ano de construção	1981	1971	1968
Área (km^2)	7,5	4,9	2,8
Volume (m^3)	88.000.000	37.000.000	15.423.000
Profundidade máxima (m)	40	18	16
Tempo de retenção hidráulica (dias)	351	356	nd
Cota do vertedouro (m)	760	837	773
Flutuação do nível do reservatório (m) (2008 - 2009)	5,71	2,54	0,70

nd= não disponibilizado - (Fonte: HDC-2000; COPASA, 2004, Mello, 2010)

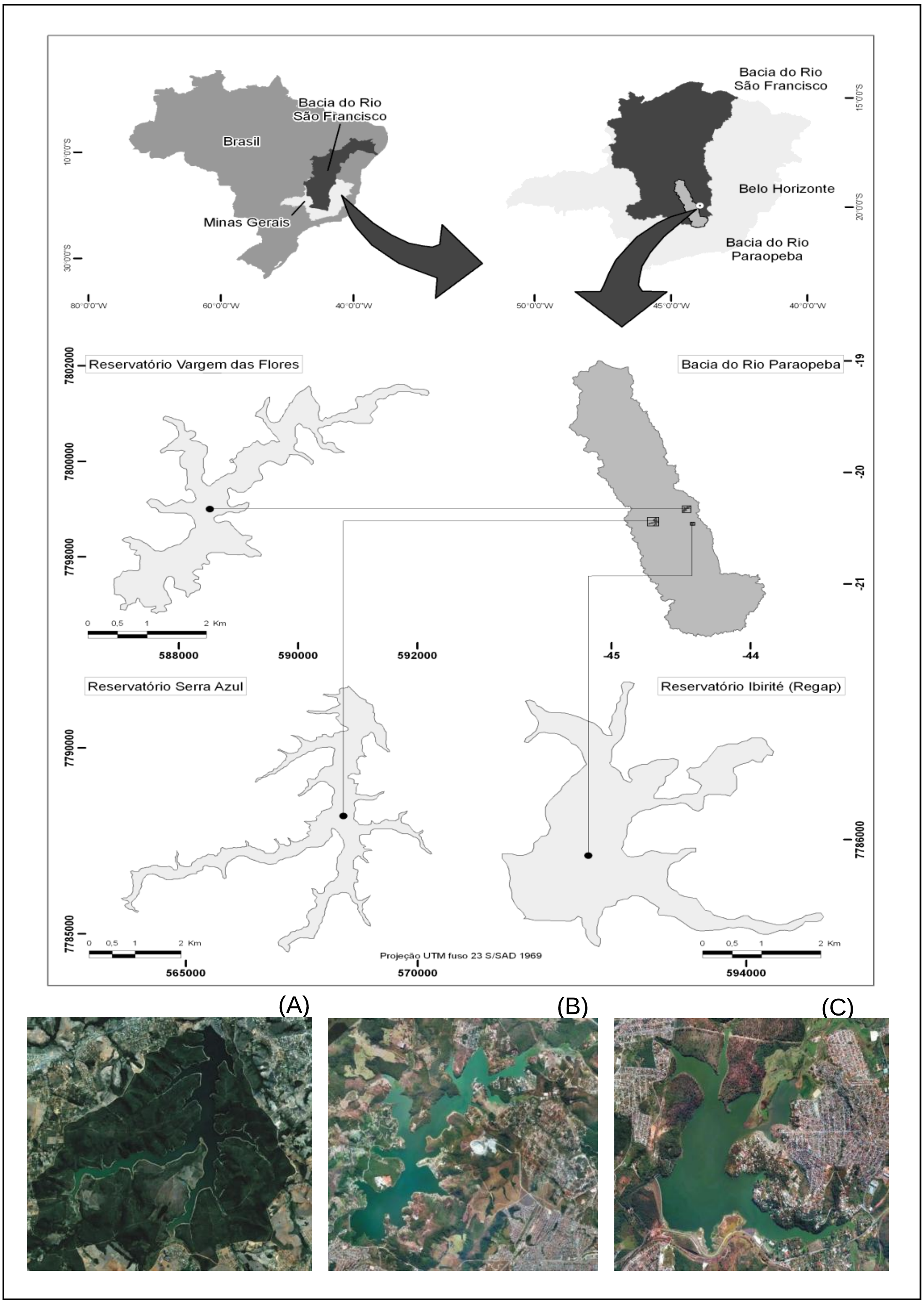

Fig. 2 Mapa com a localização dos reservatórios de Serra Azul (A), Vargem das Flores (B) e Ibirité (C), na bacia hidrográfica do Rio Paraopeba (MG).

CAPÍTULO 1

Diversity of larvae of littoral Chironomidae (Diptera: Insecta) and their role as bioindicators in urban reservoirs of different trophic levels

Sophia de Souza Morais[1], Joseline Molozzi[1], Adriana L. Viana[1],

Tales H. Viana[2] & Marcos Callisto[1]

Reservatório de Serra Azul

1- Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, ICB, Universidade Federal de Minas Gerais

2- Companhia de Saneamento de Minas Gerais-COPASA

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Diversity of larvae of littoral Chironomidae (Diptera: Insecta) and their role as bioindicators in urban reservoirs of different trophic levels

Sophia de Souza Morais, Joseline Molozzi, Adriana Lessa Viana,

Thales Heliodoro Viana & Marcos Callisto

Abstract: The Chironomidae (Diptera: Insecta) have a high species richness, with species adapted to live under widely different environmental conditions. The study of the taxonomic composition of chironomid larvae and the percentage of occurrence of deformities in mouthparts, mainly in the mentum, are used in biomonitoring programmes in order to obtain information on the levels of organic and chemical pollution of aquatic ecosystems. The objective of this study was to evaluate the abundance of chironomid larvae and to quantify the occurrence of mentum deformities in the specimens collected in three urban reservoirs with different trophic levels. The reservoirs are located in the hydrographic basin of the Paraopeba River, an affluent of the São Francisco River basin (Minas Gerais State, southeastern Brazil). The Serra Azul Reservoir is oligotrophic, the Vargem das Flores Reservoir is mesotrophic, and the Ibirité Reservoir is eutrophic. Along the littoral zone of each reservoir, 30 samples were collected during each sampling campaign. Sampling was carried out every three months for one year, with two sampling campaigns during the wet season and two during the dry season in 2008. Physical and chemical parameters measured in the water column included the water depth, Secchi depth, air and water temperature, electrical conductivity, total dissolved solids, redox potential, dissolved oxygen, pH, turbidity, Total-N, Total-P, P-ortho and chlorophyll-*a*. The chironomid larvae were identified to the genus level. The structure of the chironomid assemblages was evaluated based on taxonomic richness (24 genera), density, equitability, and diversity. The potential indicator taxa for each reservoir were established through an Indicator Species Analysis. The values for taxonomic richness (20 taxa), equitability (0.737), and Shannon-Wiener diversity (2.215) were highest in the Serra Azul Reservoir. *Fissimentum* was the indicator taxon in Serra Azul, the oligotrophic reservoir; whereas *Pelomus* was the indicator taxon in Vargem das Flores, and *Chironomus* in Ibirité. The highest percentage of mentum deformities was found during the dry season in Serra Azul (6.9%), while the lowest percentage was found during the wet season in Vargem das Flores (0.8%). The results of this study evidenced significant differences in the taxonomic composition, richness, equitability and diversity of the chironomid assemblages in these three reservoirs of different trophic levels.

Keywords: Taxonomic composition, indicator taxa, Chironomidae deformities.

Introduction

Reservoirs are artificial ecosystems, and their ecological functioning has intermediate characteristics between rivers and lakes (Tundisi et al., 1998). Reservoirs are constructed in order to provide water reserves for different purposes including the production of electricity, household and industrial supplies, transport, irrigation, and recreation (Branco & Rocha, 1977; Tundisi et al., 2008). Reservoirs are distinct landscape features, and in Brazil their surrounding areas are often the target of uncontrolled human occupation (Tundisi, 2006). Anthropogenic reservoir eutrophication leads to an increase in nutrient concentrations (nitrogen and phosphorus) and to the alteration of physical and chemical water parameters (temperature, dissolved oxygen, pH, electrical conductivity), causing reduction of the aquatic biodiversity (Camargo et al., 2005) and often cyanobacterial blooms (Costa et al., 2006; Conley et al., 2009).

Freshwater bioindicators are species, groups of species, or biological communities whose presence, density, and distribution indicate the magnitude of environmental impacts in an aquatic ecosystem and its catchment basin (Bonada et al., 2006). Biological communities reflect the ecological integrity of their ecosystems, integrating the effects of different impacting agents and providing an aggregate measure of the impact of these agents (Barbour et al., 1999). Biological indicators of water quality offer important advantages over physical and chemical parameters, since they represent environmental conditions obtained over periods of time, whereas physical and chemical data are instant measurements that reflect only the present conditions in the aquatic ecosystems (Callisto et al., 2005a).

There are many biological indicators of ecological conditions in freshwater ecosystems. The most frequently used are the benthic macroinvertebrates (Karr, 1991;

Piedras et al., 2006). These organisms are widely used in biomonitoring programmes because they directly reflect environmental changes in aquatic ecosystems and their catchments; they are sedentary, diverse, and abundant; and they have long life cycles, which allow some temporal and spatial stability (Rosenberg & Resh, 1993).

Studies of reservoir water quality using benthic macroinvertebrates have contributed to increased ecological knowledge of the communities of these aquatic ecosystems (Roque et al., 2004; Moreno & Callisto, 2006; Jorcin et al., 2009). In many Brazilian reservoirs, the benthic communities are represented by three main groups: Oligochaeta, Mollusca and larvae of Chironomidae (Pamplin et al., 2006; Jorcin & Nogueira, 2008). Chironomid larvae comprise a prominent part of the benthic macrofauna because of their high species richness and adaptability to different environmental conditions (Oliver, 1971; Coffman & Ferrington Jr., 1996; Callisto et al., 2002).

Many studies have demonstrated that the physical and chemical parameters of the water influence chironomid composition and abundance (Oliver, 1971; Botts, 1997; Helson et al., 2006; Entrekin et al., 2007). The success of this family in exploiting a wide rage of trophic conditions in aquatic ecosystems is a consequence of its great capacity for physiological adaptation, which allows the individuals to live in environments where temperature, pH, dissolved oxygen concentration, pollution, salinity, depth, and productivity are variable (Helson et al., 2006; Entrekin et al., 2007). As a result, these organisms are able to colonise many types of substrates in high densities (Berg & Hellenthal., 1992; Tokeshi, 1995; Huryn & Wallace, 2000). These characteristics make chironomids efficient organisms for the evaluation of water quality in Neotropical reservoirs (Takahashi et al., 2008).

Reports on morphological abnormalities in the heads of chironomid larvae collected in polluted environments suggest a relationship between these deformities and pollution (Lenat, 1993; Vermeulen, 1995; Janssens de Bisthoven et al., 1998; Servia et al., 2000). The deformities are reported to be more frequent in more-polluted aquatic ecosystems, and some studies have used their frequency as an indicator of severe pollution (Servia et al., 2000; Martinez et al., 2002). These deformities occur at different intensities in the antenna and mouthparts, mainly in the mentum, when the larvae are exposed to heavy metals, agricultural pesticides and fertilisers, and industrial pollutants, among others (MacDonald & Taylor, 2006; Sanseverino & Nessimian, 2008).

The objective of this study was to evaluate the taxonomic composition, distribution, and abundance of chironomid larvae, as well as to quantify the occurrence of mentum deformities in specimens collected in three reservoirs of different trophic levels. The hypothesis was that human activities in the catchment basin of a reservoir alter the composition and structure (richness, equitability and diversity) of the chironomid assemblages in the littoral zone of reservoirs, and can cause morphological deformities in the mouthpart structures of these organisms.

Because the level of degradation of a reservoir is usually related to its trophic status. we predicted that: i) the oligotrophic reservoir, well preserved and relatively unimpacted, will show higher richness and equitability than the mesotrophic and eutrophic reservoirs, which are impacted by industrial activities and by the disposal of domestic sewage; ii) a higher frequency of occurrence of morphological deformities in the mentum of chironomid larvae will be observed in the mesotrophic and eutrophic reservoirs; and iii) the mesotrophic and eutrophic reservoirs will have high nutrient contents in the water, and pollution-tolerant taxa; whereas the oligotrophic reservoir will have low nutrient contents and pollution-sensitive taxa.

Material and Methods

Study area

The study was carried out in three reservoirs located in the catchment of the Paraopeba River, an affluent of the São Francisco River basin (Minas Gerais State, Brazil) (Fig. 1).

The Ibirité Reservoir (19° 07’ 00”-20° 02’ 30” S and 44° 07’ 30-44° 05’ 00” W) is formed by the influx of the Pintado and Retiro do Onça rivers. It is affected by intense human impacts such as the disposal of domestic sewage and the presence of unorganised human settlements in its surroundings. As a consequence, this ecosystem shows advanced artificial eutrophication (Callisto et al., 2005b; Moreno & Callisto, 2006). The reservoir has a surface area of 2.8 km^2, a volume of 15,423.000 m^3 and a mean depth of 16 m (Rodrigues, 2004).

The Vargem das Flores Reservoir (19° 53’ 30’’-19° 55’ 25” S and 44° 07’ 22” and 44° 10’ 59” W) is fed by Betim Creek. It has moderate human occupancy in its surroundings and is, together with the Serra Azul Reservoir, one of the main sources of water supply for the Belo Horizonte metropolitan region. It is a mesotrophic ecosystem, with a surface of 5.5 km^2, a volume of 44,000.000 m^3, a mean depth of 6 m, and a maximum depth of 18 m (COPASA, 2004).

The Serra Azul Reservoir (19° 54’ 09” - 20º 00’ 52’’ S and 44° 23’ 16”-44° 30’ 20” W) is fed by Juatuba Creek. It is located on the boundary between the Juatuba and the Mateus Leme municipalities. It is an oligotrophic ecosystem, with a surface of 8.9 $km^{2,}$ a volume of 93,000.000 m^3 and a maximum depth of 40 m (COPASA, 2004). The sediment has high manganese levels due to its geomorphologic origin (Martins, 1996).

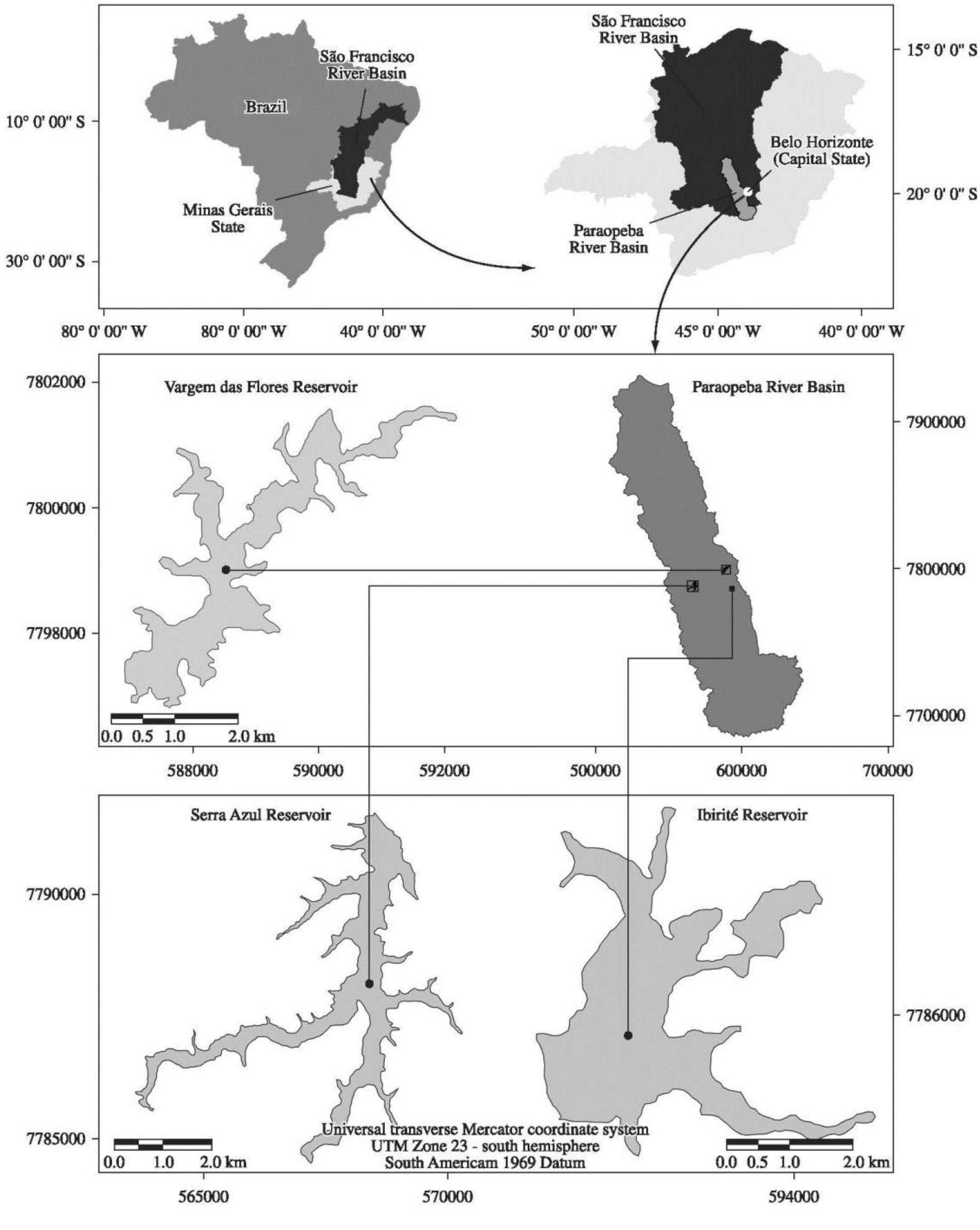

Fig. 1 Map of the Vargem das Flores, Serra Azul and Ibirité reservoirs located in the basin of the Paraopeba River (MG).

Field methods and laboratory analyses

The reservoirs were sampled every three months in 2008, during the dry season (June and September) and during the wet season (March and December). Along the littoral zone of each reservoir, 30 samples were collected, using an Eckman-Birge (0.0225 m^2) sampler.

The samples were deposited in plastic bags and transported to the laboratory, where they were washed on sieves of 1 mm and 0.5 mm meshes (Larsen et al., 1991). Sub-surface water samples were collected using Van Dorn bottles, for the measurement of the physical and chemical parameters. The physical and chemical parameters of the surface water (pH, temperature, dissolved oxygen, electrical conductivity and turbidity) were measured in situ, using a multi-analyser and portable apparatus (YSI). The Secchi disc was used to evaluate the depth of the trophic zone. For the measurements of total nitrogen, total phosphorus, and orthophosphate, 30 water samples were collected from each reservoir and transported to the laboratory in refrigerated polyethylene bottles. These measurements were performed according to the Standard Methods for the Examination of Water and Wastewater (APHA, 1992). In order to analyse the chlorophyll-*a* content, 500 mL of reservoir water was filtered through Millipore AP40 filters. After filtration, the filters were manually macerated and extracted with 90% acetone following the procedure described by Golterman et al. (1978). The trophic state index (Carlson, 1977), which uses the Total-P values, the chlorophyll-*a* values, and the Secchi disc values, were used to assess the trophic level of the reservoirs. Values of this index equal to or less than 20 indicate ultra-oligotrophy, values between 21 and 40 indicate oligotrophy, values between 41 and 50 evidence mesotrophy, values between 51 and 60 evidence eutrophy, and values equal or greater than 61 indicate hyper-eutrophy.

Chironomid larvae

The chironomid larvae were treated with a 10% lactophenol solution and identified under a microscope (400x) with the aid of taxonomic keys (Trivinho-Strixino & Strixino, 1995; Epler, 2001). The occurrence of morphological deformities in the mentum was recorded and counted. All chironomids collected were analysed. The lack or excess of teeth, asymmetry, fusion, tooth malformation, and combinations of these characteristics were considered deformities. We considered the presence or absence of deformities, but did not calculate their frequencies (Dickman et al., 1992).

Sampling stations that provided samples with deformity frequencies equal to or less than 3% were considered natural, between 3 and 6% were considered altered, and impacted when the frequency of deformities exceeded 6% (Burt et al., 2003).

Data analyses

The Shannon-Wiener diversity index, Pielou's equitability index (Magurran, 1988), organism density (individuals/m^2), and taxonomic richness (total number of taxa in each sample) were calculated in order to evaluate the structure of the chironomid assemblages.

A variance analysis (ANOVA) (Software Statistica for Windows 5.1) of the data on the composition of the chironomid assemblages was used to evaluate if there were significant differences among the three reservoirs.

A cluster analysis (Software Primer 6 Beta, 2004) was performed in order to assess the similarity in the taxonomic composition of the assemblages found in the three reservoirs. The Bray-Curtis index and an UPGMA (Unweighted Pair Group Method with Arithmetic Mean) were used as the amalgamation method.

Using the three reservoirs as the groups to be indicated, an indicator species analysis (Dufrêne & Legendre, 1997) using the PC-Ord software (version 3.11, 1997) was carried out in order to establish the indicator taxa for each reservoir. The taxa that showed p-values below <0.05 in a randomisation Monte Carlo test (10.000 randomisations) were considered to be indicators for one or two reservoirs.

Results

In total, 9981 organisms were collected, of which chironomid larvae represented 25.95%. A total of 1697 individuals were collected in Serra Azul Reservoir (32.70% Chironomidae), 3853 individuals in Vargem das Flores Reservoir (15.21% Chironomidae), and 4431 individuals in Ibirité Reservoir (32.70% Chironomidae).

A total of 2590 Chironomidae larvae were collected in the three reservoirs. The specimens belonged to 23 genera and two subfamilies: Subfamily Tanypodinae: 2 Pentaneurini genera, 2 Procladiini genera, 1 Coelotanypodini genus, and 1 Tanypodini genus; Subfamily Chironominae: 13 Chironomini genera, 3 Tanytarsini genera, and 2 Pseudochironomini genera. Members of Chironominae represented 64.05% of the individuals, whereas the Tanypodinae comprised 35.95%. The 1449 individuals found in Ibirité Reservoir belonged to 11 genera, with *Chironomus* being the most abundant (31.43%), followed by *Coelotanypus* (20.63%), *Aedokritus* (16.66%), *Tanypus* (14.71%), and *Tanytarsus* (9.66%). The 586 individuals collected in Vargem das Flores Reservoir belonged to 16 genera, with *Coelotanypus* being the most abundant (34.27%), followed by *Aedokritus* (30.59%), *Pelomus* (9.28%), *Djalmabatista* (7.56%), and *Tanypus* (4.76%). The 555 individuals collected in Serra Azul Reservoir belonged to 20 genera, with *Fissimentum* being the most abundant (19.75%), followed by *Tanypus* (17.07%), *Coelotanypus* (15.96%), *Procladius* (11.03%), *Djalmabatista* (10.90%), and *Polypedilum* (8.37%) (Table 1).

Regarding the chironomid larvae, Serra Azul showed the highest values of taxonomic richness (20 taxa), Pielou's equitability (0.737), and Shannon-Wiener diversity (2.215), and the lowest total density (1.494.81 ± 2,228.16 individuals/m^2). Vargem das Flores showed intermediate values of taxonomic richness (16 taxa), equitability (0.655), diversity (1.817), and total density (1.812.22 ± 3,945.36 individuals/m^2). The lowest taxonomic richness (10 taxa), equitability (0.614), diversity (1.473) values, and the highest total density (4.235.75 ± 8,775.31 individuals/m^2) value were all recorded for Ibirité (Table 1).

Table 1. Median and standard deviation of Chironomidae larvae collected at the Vargem das Flores, Serra Azul and Ibirité reservoirs during 2008.

	Ibirité	**Vargem das Flores**	**Serra Azul**
TANYPODINAE			
Coelotanypodini			
Coelotanypu Kieffer,1913	20.63	34.27	15.96
Pentaneurini			
Ablabesmyia Johannsen, 1905	0.97	0.85	5.90
Labrundinia Fittkau, 1962			0.12
Procladiini			
Djalmabatista Fittikau, 1968		7.56	10.90
Procladius Skuse, 1889		0.74	11.03
Tanypodini			
Tanypus Meigen,1803	14.71	4.76	17.07
CHIRONOMINAE			
Chironomini			
Aedokritus Roback, 1958	16.66	30.59	0.37
Beardius Reiss & Beck, 1985 et Sublette		0.10	
Chironomus Meigen, 1803	31.43	2.25	3.82
Cladopelma Kieffer, 1921		0.20	0.25
Cryptochironomus Kieffer, 1918			0.12
Dicrotendipes Kieffer, 1913	0.04	0.10	
Fissimentum Cranston & Nolte, 1996		1.25	19.75
Goeldichironomus Fittkau, 1965	1.18	2.23	
Pelomus Kieffer,1921	2.91	9.28	1.19
Lauterboniella Thienemann Bause, 1913			1.19
Paralauterboniella Lenz, 1941			0.12
Polypedilum Kieffer, 1913		4.27	8.37
Stenochironomus Kieffer, 1919			0.12
Pseudochironomini			
Manoa Fittikau			0.45
Pseudochironomus Mallock, 1915			0.57
Tanytarsini			
Tanytarsus Kieffer, 1921	9.66	1.14	0.74
Não identificados	1.81	0.41	1.96
Taxonomic Richness	11	16	20
Diversity	1.473	1.817	2.215
Equitability	0.614	0.655	0.737
Total Density (ind/m^2)	4235.75 ± 8775.31	1812.22 ± 3945.36	1494.81 ± 2228.16

The values for chironomid taxonomic richness ($F_{2.87}$ = 4.60, p = 0.01), Pielou's equitability ($F_{2.86}$ = 4.73, p = 0.01), and Shannon-Wiener diversity ($F_{2.87}$ = 3.728, p = 0.02) among the three reservoirs were significantly different. On the other hand. the total density values for the three reservoirs were not significantly different ($F_{2.87} = 1.63$, $p = 0.20$). The taxonomic composition was significantly different among the reservoirs when the dry and the wet seasons were compared ($F_{153.38} = 3.177$, $p = 0.0016$).

The analysis of indicator species showed *Chironomus* as the indicator taxon for Ibirité Reservoir, with an indicator value of 81; *Pelomus* as the indicator taxon for Vargem das Flores Reservoir, with an indicator value of 31; and *Fissimentum* as the indicator taxon of Serra Azul Reservoir, and an indicator value of 75. The cluster analysis showed a higher similarity between Ibirité and Vargem das Flores reservoirs, which were separated from Serra Azul Reservoir (Fig. 2).

Despite the importance of identification of genera belonging to the subfamily Tanypodinae for the study of the taxonomic composition of the chironomid assemblages, deformities in mouthparts were found only among larvae belonging to the subfamily Chironominae. During the dry season in Serra Azul Reservoir, the occurrence of morphological deformities exceeded 6% (6.9%). In this reservoir during the wet season, and in Ibirité and Vargem das Flores reservoirs, the percentages of morphological deformities were less than 6% (Table 2).

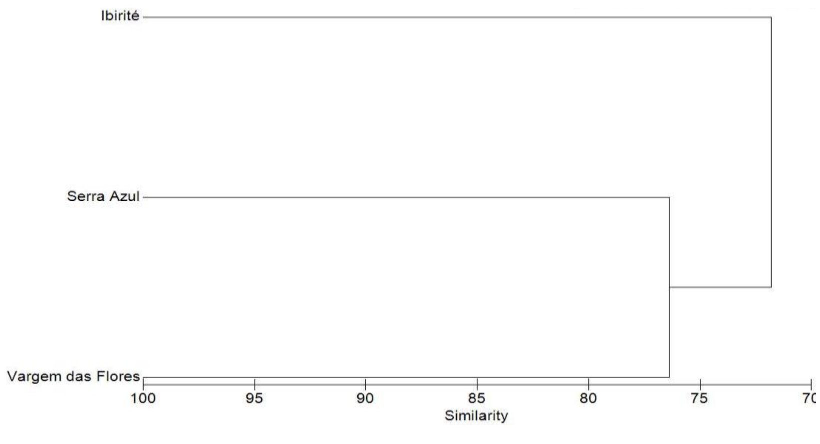

Fig. 2 Similarity dendrogram for the taxonomic composition of Chironomidae found in Serra Azul, Vargem das Flores and Ibirité reservoirs.

Deformities in the mentum of chironomid larvae were found in the genera *Aedokritus, Chironomus, Fissimentum* and *Polypedilum*, *Aedokritus* individuals with deformities were found in Ibirité and Vargem das Flores. *Chironomus* larvae with mentum deformities were found in all three reservoirs. Individuals belonging to the genera *Fissimentum* and *Polypedilum* that displayed morphological deformities were only found in Serra Azul.

Table 2. Percentage of total found in mentum deformites of chironomid larvae belongin to the subfamily Chironominae and total percentage of deformities found in each gender in reservoirs Ibirité, Vargem das Flores and Serra azul during the dry and rainy seasons in 2008.

	Ibirité		**Vargem das Flores**		**Serra Azul**	
	Dry	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**
CHIRONOMINAE	1.54	1.48	2.46	0.80	6.90	3.29
Chironomini						
Adokritus Roback, 1958	3.53	2.80	4.21			
Chironomus Meigen, 1803	1.50	1.15		16.66	16.66	
Fissimentum Cranston & Nolte, 1996					4.76	
Polypedilum Kieffer, 1913					40.00	7.69

The results of the physical and chemical analyses (Table 3) showed that the highest value for Total-P was found in Ibirité during the dry season (229.19 mg/L), and the lowest value was observed in Serra Azul during the wet season (19.40 mg/L). Intermediate values were found in Vargem das Flores (21.41 mg/L during the dry season and 24.90 mg/L during the wet season). The highest chlorophyll-*a* concentration was found in Ibirité during the dry season (90.08 µg/L), while Serra Azul showed low concentrations (2.13 µg/L) throughout the year and Vargem das Flores showed intermediate concentrations during both the dry (2.67 µg/L) and the wet (3.55 µg/L) seasons. The highest electrical conductivity was recorded in Ibirité during the dry season (393.0 µS/cm), while the lowest value was found in Serra Azul during the dry season

(29.35 µS/cm), and intermediate values were found in Vargem das Flores during the dry (150.43 µS/cm) and wet (134.0 µS/cm) seasons.

The results found for the trophic status of the reservoirs indicated oligotrophy in Serra Azul (39.85), mesotrophy in Vargem das Flores (46.92), and eutrophy in Ibirité (69.77).

Table 3. Median and standard deviation physical and chemical characterisation of the Vargem das Flores, Serra Azul and Ibirité reservoirs during the wet and rainy seasons in 2008.

	Ibirité		**Serra Azul**		**Vargem das Flores**	
	Dry	**Rainy**	**Dry**	**Rainy**	**Dry**	**Rainy**
Depth (m)	2.6 ± 2.3	2.8 ± 2.88	4.37 ± 3.53	5.47 ± 3.53	4.40 ± 2.95	3.81 ± 2.01
Secchi (m)	0.51 ± 0.37	0.48 ± 0.16	2.59 ± 0.76	1.65 ± 0.47	1.21 ± 0.51	0.99 ± 0.31
Air Temperature (ºC)	27.03 ± 2.6	27.8 ± 2.77	22.8 ± 1.14	29.4 ± 2.2	25.77 ± 3.11	27.5 ± 1.98
Water Temperature (ºC)	25.2 ± 2.2	27.5 ± 1.72	23.97 ± 1.14	29.54 ± 0.96	27.16 ± 2.8	28.3 ± 1.80
pH	8.2 ± 0.99	7.6 ± 0.90	7.50 ± 0.27	7.28± 0.39	8.29 ± 0.29	7.9 ± 0.58
Electrical Conductivity (µS/cm)	393.0 ± 63.71	203.2 ± 145.30	29.35 ± 1.33	33.37 ± 11.57	150.43 ± 19.61	134.0 ± 33.27
TDS (mg/L)	336.33 ± 45.99	208.54 ± 30.14	21.60 ± 1.45	22.05 ± 6.96	120.69 ± 31.61	98.6 ± 12.3
Redox (mV)	62.41 ± 119.35	104.66 ± 93.3	152.61 ± 32.91	187.5 ± 67.47	113.7 ± 16.97	208.33 ± 44.37
Turbidity (UNT)	59.52 ± 88.83	99.55 ± 1.54	3.44 ± 17.19	13.12 ± 17.19	39.14 ± 33.36	47.34 ± 41.06
Dissolved Oxygen (mg/L)	7.19 ± 2.20	7.88 ± 2.38	7.40 ± 0.96	7.40 ± 0.90	7.60 ± 1.47	7.44 ± 1.51
Chlorophyll-*a* (µg/L)	90.08 ± 97.30	2.52 ± 27.43	2.13 ± 1.80	2.13 ± 2.08	2.67 ± 1.79	3.55 ± 2.35
Total-N (mg/L)	0.27 ± 0.29	0.29 ± 0.16	0.04 ± 0.02	0.05 ± 0.08	0.14 ± 0.15	0.36 ± 0.30
Total-P (mg/L)	229.19 ± 421.70	130.93 ± 180.71	22.78 ± 22.78	19.40 ± 17.80	21.41 ± 14.74	24.90 ± 19.81
P-orto (mg/L)	35.37 ± 161.03	14.72 ± 25.20	5.45 ± 1.77	5.56 ± 1.60	7.69 ± 3.40	8.08.72 ± 4.54

Discussion

The taxonomic composition of Chironomidae in the three reservoirs was typical of lentic ecosystems, including *Fissimentum, Goeldichironomus* and *Cladopelma* (Strixino & Trivinho-Strixino, 1998; Leal et al., 2004).

Serra Azul Reservoir was the most diverse, in terms of both taxonomic richness (20 taxa) and Shannon-Wiener diversity (2.215). This reservoir had the genus

Fissimentum, which is common in good-quality freshwaters (Leal et al., 2004), as its indicator taxon. In contrast, the chironomid assemblage in Ibirité Reservoir had the genus *Chironomus* as the indicator taxon, and showed low levels of taxonomic richness (11 taxa) and diversity (1.473) compared to Serra Azul Reservoir.

The genus *Chironomus* is characterised by its tolerance to pollution and high organic-matter concentrations, thus being typical of impacted ecosystems (Devái, 1988; Marques et al., 1999; Helson et al., 2006). Vargem das Flores Reservoir showed intermediate levels of taxonomic richness (16 taxa) and diversity (1.817) compared to the other two reservoirs, and had as an indicator taxon the genus *Pelomus*, which is typical of clean environments (Simpson & Bode, 1980) and lives on sand substrata of lentic littoral zones (Strixino & Trivinho-Strixino, 1998).

The high abundance of individuals of the genus *Fissimentum* in Serra Azul is due to the fact that members of this genus are common in aquatic ecosystems where there are fluctuations of the water level (Cranston & Nolte, 1996), and are typical of lentic ecosystems that have good ecological potential (Leal et al., 2004).

Seasonal differences in the taxonomic composition and density could be explained by the different amounts of allochthonous material entering these systems. Increased input of allochthonous matter produces a decrease in organism density (Higuti & Takeda, 2002).

The high percentage of deformities found in individuals of the subfamily Chironominae in Serra Azul (>6%) may be a response to the high concentrations of manganese (>0.1 mg/L) present in the sediment of this reservoir, due to the geological nature of the underlying matrix (Martins, 1996). The manganese content in this reservoir exceeds the acceptable level (CONAMA Resolution No. 357 of March 17th, Brasil, 2005) for special-class aquatic ecosystems. High concentrations of this heavy metal in the sediment of aquatic ecosystems can cause the development of morphological deformities in chironomid larvae (Janssens de Bisthoven et al., 2005). Toxic contaminants can influence the presence of chironomids because these insects depend on microhabitat structure and physico-chemical conditions; and contaminants may also cause malformations in the larvae (Nazarova et al., 2004).

Reservoir (11 taxa) could be due to the high concentrations of chlorophyll-*a*, Total-P, and Total-N observed in the water column, which are characteristic of eutrophic ecosystems (Camargo et al*.,* 2005). In theory, an oligotrophication of the reservoir with the consequent reduction of these parameters would favour an increase of its taxonomic richness, allowing the number of taxa to reach the levels found in Vargem das Flores (16 taxa) and Serra Azul (20 taxa).

The hypothesis of the study was partially corroborated. As expected, the results evidenced significant differences in taxonomic composition, richness, equitability, and diversity among the three reservoirs. Pollution-tolerant taxa were recognised as indicators in Ibirité and Vargem das Flores reservoirs, whereas a pollution-sensitive taxon was recognised as an indicator in Serra Azul. In addition, a low similarity in composition and distribution of the chironomid larvae was observed for Serra Azul compared to the other two reservoirs. High percentages of morphological deformities were found only in the oligotrophic reservoir, and were probably due to the high manganese content in the sediments.

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CAPÍTULO 2

Size-mass relationships of *Melanoides tuberculatus* (Thiaridae, Gastropoda) in an eutrophic reservoir

Eduardo Carvalho Silva, Joseline Molozzi & Marcos Callisto

Reservatório de Ibirité

Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, ICB, Universidade Federal de Minas Gerais

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Size-mass relationships of *Melanoides tuberculatus* (Thiaridae, Gastropoda) in an eutrophic reservoir

Eduardo Carvalho Silva, Joseline Molozzi & Marcos Callisto

Abstract: This study evaluated the relationships of certain allometric measurements in *Melanoides tuberculatus* Muller, 1774, in order to develop a statistical model to estimate the biomass of this mollusc species. We measured the total length and aperture of 70 shells. These measurements were correlated with the biomass values to construct exponential and power-function models, and both models showed high coefficients of determination. The exponential model was the better biomass predictor, with a coefficient of determination over 93%. These proposed models may be an effective tool to determine the biomass of *M. tuberculatus* in eutrophic Brazilian reservoirs.

Keywords: Allometry, dry-mass, eutrophication, molluscs.

Introduction

Reservoirs are artificial ecosystems that functions mainly to generate electricity and supply water for domestic and industrial processes. From the ecological point of view, reservoirs produce considerable alterations in aquatic ecosystems due to changes in the time of water residency, habitat fragmentation and exotic species invasion (Tundisi, 2008; Agostinho et al., 2005). Eutrophication is also a serious threat, since the excessive growth of algae resulting from enrichment of the water, mainly with phosphorus and nitrogen, can modify the biodiversity and the distribution of organisms in these environments (Figueirêdo et al., 2007).

In tropical countries, both native mollusc species and exotic invasive species such as *Melanoides tuberculatus* Muller, 1774 occur in reservoirs. The Afro-Asian *M. tuberculatus* was originally reported in Brazil in 1967 in Santos (São Paulo), and spread to Brasília (Distrito Federal) and the states of Rio de Janeiro, Goiás, Paraíba and Espírito Santo (Fernandez et al., 2003; Vaz et al., 1986). *Melanoides tuberculatus* was first reported in the state of Minas Gerais in 1986, from Pampulha Reservoir in the city of Belo Horizonte (Carvalho, 1986; Freitas et al., 1987). In a study begun in September of that same year at Soledade Lake in the Ouro Branco region, but not published until 1994, some individuals were also found (Silva et al., 1994). In 1996, the species was reported from Dom Helvécio Lake in the Rio Doce State Park (De Marco Jr, 1999).

This invasive mollusc threatens the biodiversity of ecosystems where it is found, and it also represents a risk for human health since it hosts the trematodes *Paragonimus westermani* Kerbert, 1878 and *Clornorchis sinensis* Looss, 1907 species endemic to Asia that are able to parasitize humans (Souza & Lima, 1990). The species is an r strategist, with parthenogenetic reproduction and the potential to maintain high population densities for long periods of time. It is easily transported and highly adaptable, establishing on all kinds of substrates (Pointier et al., 1993).

Because of all these reasons, *M. tuberculatus* is found in many aquatic ecosystems with different levels of pollution and ranging from oligotrophic to hypereutrophic (Rocha-Miranda & Martins-Silva, 2006; Callisto et al., 2005). Since *M. tuberculatus* is a potential competitor of native planorbid species that host human parasites, it has been introduced into the American continent for purposes of biological control (Guimarães et al., 2001).

In studies on benthic aquatic insects, biomass estimates provide information to estimate growth, production, and feeding ecology (Benke et al., 1999; Johnston & Cunjak, 1999). Similarly, information on biomass can be used to support inferences about the adaptation of *M. tuberculatus* in eutrophic environments and its effect on other species of molluscs (Pointier & Augustin, 1999).

However, biomass estimation takes time and is prone to errors, especially when the organisms are preserved. Preservatives can produce significant losses of biomass and most are toxic (Wetzel et al., 2005). Because of these and other problems, that many studies on insects have used the size of body structures as biomass predictors (Genkai-Kato & Miyasaka, 2007; González et al., 2002; Cressa, 1999). However, the relationships between size and weight must be used cautiously since they may not account for environmental and geographical variations and are not exact for every taxon, i.e., there is a different relationship for each species (Johnston & Cunjak, 1999).

Since *M. tuberculatus* was recorded in Brazilian waters, few studies have assessed the spread of this invasive mollusc in this country. The status of the species is still poorly known, as is its distribution and ecology (Fernadez et al., 2003). The size structure of a population of *M. tuberculatus* in a eutrophic reservoir might be an important tool to aid in the management of this alien species in southeastern Brazil.

The objective of the present study was to establish, through allometric relationships, a statistical model to estimate the biomass of *M. tuberculatus* populations.

Material and Methods

The Ibirité Reservoir (19°07’00”-20º02’30”S, 44°07’30-44º07’30”W) is fed mainly by the Ibirité River. The dam was construted in 1967 in order to supply water for the Gabriel Passos refinery (REGAP), one of the Petrobras refineries constructed along highway BR-381. The reservoir has a surface area of 2.8 km^2, a volume of 15.423.000 m^3 and a mean depth of 16 m (Garcia et al., 2009) (Fig. 1).

The reservoir has undergone rapid artificial eutrophication because of human impacts in its drainage basin. This eutrophic state is characterized by high primary production, algal bloom episodes, and the presence of aquatic macrophytes (Moreno & Callisto, 2006). During the study period the mean depth at the sampling stations was 2.6 m (± 1.65). Mean dissolved oxygen concentrations were about 7.3 mg/l (± 0.68), with a mean temperature of 27.83 °C (± 0.52), a slightly basic pH (7.4 ± 0.32) and high electrical conductivity (237.6 µS cm^{-1} ± 23.16).

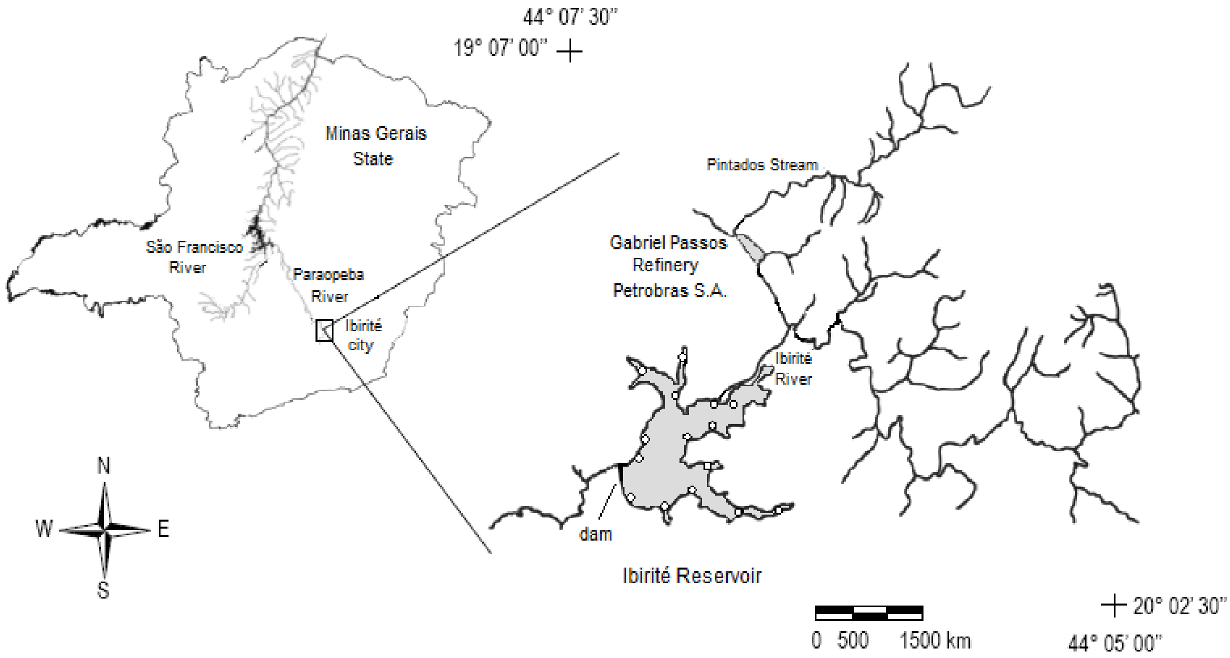


Fig. 1 Distribution of sampling points in the Ibirité Reservoir, in the basin of the Paraopeba River, Minas Gerais. Modified from Garcia et al., 2009.

The molluscs were collected from the sediments during April 2009, by means of an Ekman-Birge sampler with a sampling area of 0.0225 m^2 (Freitas et al., 1987). In all, 15 sampling points were selected along the shore of the reservoir, and three replicates were taken at each point (Fig. 1). The collected material was placed in plastic bags and transported, with no the added of preservatives, to the Laboratório de Ecologia de Bentos, Universidade Federal de Minas Gerais. At the laboratory, the samples were washed on sieves (0.5 mm mesh size) to separate the molluscs, on the same day of their collection.

The measurements of the total length of the shell (from the vertex to the farthest point on the opposite end) and the shell aperture were taken using a Vernier caliper with a precision of 0.05 mm. Each snail was weighed to obtain the total wet weight. The specimens were dried in the oven at 60°C until they reached a constant weight, approximately 48 hours, and then incinerated in a furnace at 500°C for four hours in order to estimate the weight of the mineral fraction. The biomass was obtained from the difference between the total dry weight and the weight of the mineral fraction. The results were expressed in milligrams and correlated with the length and shell aperture measurements in millimeters, in order to establish the relationship between weight and length (Elkarmi & Ismail, 2007).

Dispersion diagrams were constructed, correlating the biomass (B) with the measurements of total length (TL) and shell aperture (SA). The method of least squares was used to produce the regression models. The validity of the equations was based on the regression significance (p), the coefficient of determination (R^2) and on the analysis of the residues (Vieira, 2003).

Results

The relations between the measurements and the biomass of 70 individuals were used. The shells had lengths varying from 3.2 to 18.95 mm, with a mean of 10 mm (±4.62); the shell openings varied from 0.75 to 5.9 mm, with a mean of 2.8 mm (± 1.47); and the biomass varied from 0.4 to 22 mg, with a mean of 5.76 mg (± 5.91).

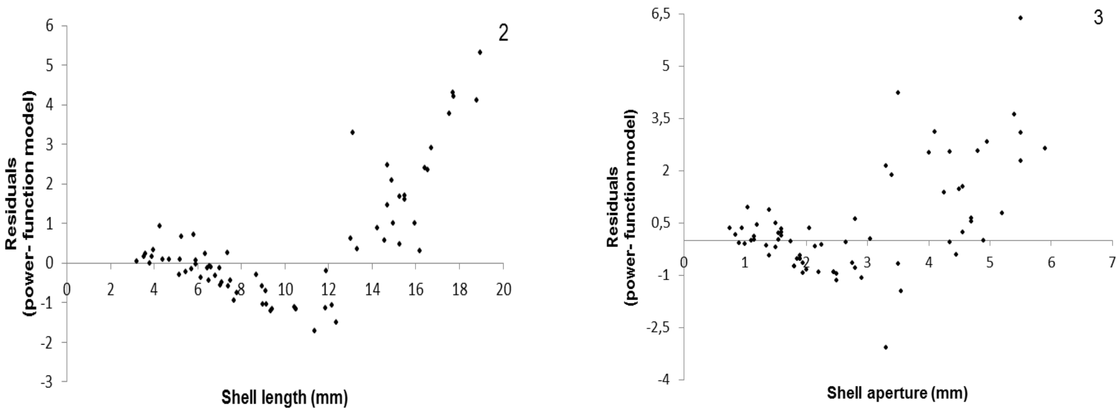
The exponential and power-function models presented significant correlations ($p < 0.001$). The coefficients of determination of the regressions (R^2) varied from from 0.9206 to 0.9685 (Table 1). The results evidenced that the shell length was the measurement that produced the highest coefficient of determination in the regression analyses of the exponential and power function models (R^2), 0.9685 and 0.9414, respectively.

Table 1. Regression equations and values found for each variable. The biomass is represented by B, and the shell total length (TL) and shell aperture (SA) of *M. tuberculatus* are represented by L. (a and b) regression constants, (R^2) coefficient of determination, (*e*) Euler's number (2.718).

Function	Equation	Conversion	A	b	R²
Exponential	$B = a.e^{(b \cdot L)}$	B→TL	0,3058	0,2365	0,9685
		B→SA	0,4172	0,7332	0,9353
Potential	B = a.Lb	B→TL	0,0287	2,1637	0,9414
		B→SA	0,5982	1,9136	0,9206

Discussion

The models showed high coefficients of determination, explaining from 92 to 96% of the biomass variation as a function of the measurements used. The coefficients of determination of the exponential model were slightly higher than the coefficients of the power-function model, showing that the exponential model better explained the data variation. The power-function model showed an irregular dispersal pattern of residuals for the measurements of both shell length and aperture (Figs 2 and 3) (Vieira, 2003).



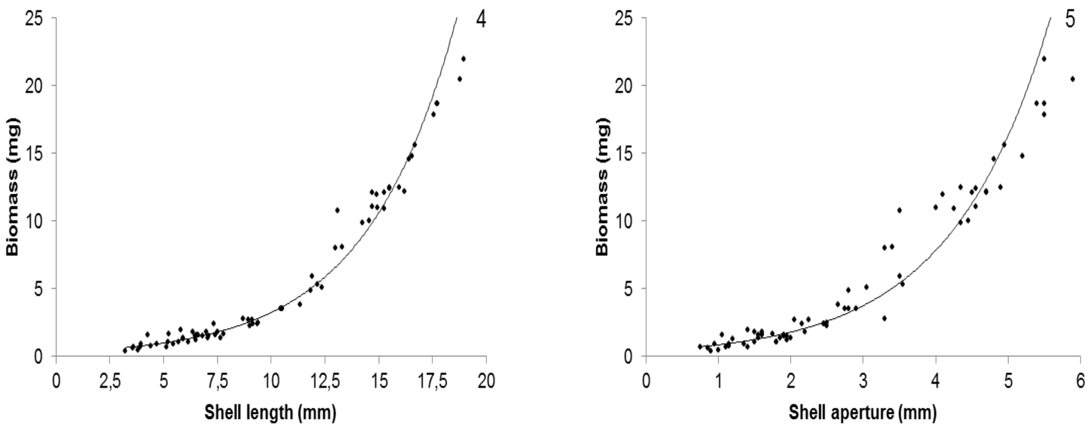
Figs. 2-3 Graphs of residual versus shell length (2) and residuals versus shell aperture (3) of the power-function model.

In practice, the interpretation of the exponencial model shows that *M. tuberculatus* has rapid growth, is well adapted to the eutrophic environment, and probably exerts competitive pressure on other species of molluscs, meeting the requirements of an invasive species (Elkarmi & Ismail, 2007; Everett, 2000; Pointier et al., 1993). Previous studies have used these relationships to propose equations for the estimation of the growth rate and/or secondary production of benthic aquatic insects (Genkai-Kato & Miyasaka, 2007; González et al., 2002; Cressa, 1999).

Some of the specimens collected were excluded from this study because of significant losses of the shell apex. A study in Pampulha Reservoir found it difficult to obtain allometric relationships because each individual had lost on average 20.53% of its estimated theoretical length, with larger individuals having lost more of the shell (Freitas et al., 1987). This breakage may occur due to factors such as water temperature, pH, total hardness, alkalinity, availability of calcium ions, and dissolved salts. These factors can influence the demineralization process and consequently lead to the loss of the entire shell (Oronsaye, 2002; Lanzer & Shafer, 1988).

Individuals with shell length and aperture greater than 10 and 3 mm, respectively, showed greater biomass variations as a function of their size (Figs 4 and 5). This may be related to the onset of sexual maturity and the stage of the reproductive period, since embryos can remain in the brood pouch for three to five months before being released (Pointier et al., 1993; Dudgeon, 1986).

Variations in the relationship between size and biomass in populations of a species from different geographical regions can be caused by differences in the physical and chemical factors and by fluctuations in their environmental conditions (González et al., 2002). This highlights the importance of selecting geographical regions with similar characteristics and preferably specific taxa, when carrying out studies that involve these kind of regressions. On the other hand, a study by Genkai-Kato & Miyasaka (2007) showed that seasonal variations did not significantly affect these relationships.



Figs. 4-5 Graphs of exponential regression model for biomass/shell length (4) and biomass/shell aperture (5) of *M. tuberculatus* in the Ibirité Reservoir, Minas Gerais (n = 70). The equation for the exponential model is: $B = a \cdot e^{(b \cdot L)}$.

In addition, the regression equations must be constructed using the data from fresh individuals, since preserved individuals can lose as much as 73.8% of their biomass during the first weeks, and this may lead to underestimation of the weight. The choice of preservative, ethanol or formaldehyde, does not seem to have a significant effect on loss of biomass, since both substances produce a large loss, mainly during the first three weeks following preservation (Wetzel et al., 2005).

The models presented here can be used to determine the biomass of *M. tuberculatus*. The exponential model better described the relationship between size and biomass and the biological characteristics of the species. Because of the presence of numerous individuals lacking the shell apex, the use of the shell aperture measurement is a good alternative for the construction of the models to describe snail populations in Brazilian eutrophic reservoirs.

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CAPÍTULO 3

Diversidade de habitats físicos e sua relação com macroinvertebrados bentônicos em reservatórios urbanos em Minas Gerais

Joseline Molozzi[1], Juliana S. França[1], Thiago L. A. Araujo[1], Tales H. Viana [2],
Robert M. Hughes[3] & Marcos Callisto[1]

Reservatório de Vargem das Flores

1- Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, ICB, Universidade Federal de Minas Gerais, Brasil
2- Companhia de Saneamento de Minas Gerais, COPASA, Brasil
3- Department of Fisheries & Wildlife, Oregon State University, Corvallis, Oregon, EUA

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Diversidade de habitats físicos e sua relação com macroinvertebrados bentônicos em reservatórios urbanos em Minas Gerais

Joseline Molozzi, Juliana S. França, Thiago L. A. Araujo, Tales H. Viana, Robert M. Hughes & Marcos Callisto

Abstract: We assessed the effects of physical habitat structure on benthic macroinvertebrate communities in 3 reservoirs: Serra Azul (SA), Vargem das Flores (VF) and Ibirité (IB). We sampled physical and chemical habitats and benthic macroinvertebrates at 30 systematic sites in each reservoir. In SA, the dominant taxa out of 12 taxa were *Melanoides tuberculatus* Müller, 1774 (15.2%) and Chaoboridae (63.8%). In VF, the dominant taxa out of 11 were *M. tuberculatus* (34.2%) and Oligochaeta (33.6%). In IB, *M. tuberculatus* (91.2%) and Chaoboridae (6.27%) dominated and we collected only 7 taxa. In SA, benthos were significantly influenced by the deciduous shrub understory and bank angle. In IB and VF, the significant physical habitat variables were land cover, human influence, and aquatic macrophytes. We conclude that human disturbances reduced terrestrial vegetation cover, riparian and littoral physical habitat structural complexity, and water quality. Those changes, in turn, reduced the taxonomic richness of the benthic macroinvertebrate communities.

Keywords: Benthic fauna, habitat integrity, semi-lentic ecosystems, benthic bioindicators.

Introdução

A ocupação desordenada de bacias hidrográficas gera impactos e deteriora a qualidade das águas, alterando sua disponibilidade, embora varie com a organização social em uma dada região (Callisto et al., 2005). Atividades antrópicas nas áreas de entorno contribuem para o assoreamento de reservatórios urbanos, reduzindo a heterogeneidade de substratos, diminuindo a disponibilidade de habitats para a biota, com consequente perda de biodiversidade (Allan, 2004). As propriedades físicas e químicas na água, juntamente com as características no entorno dos reservatórios, influenciam a distribuição dos organismos aquáticos. Assim, a redução da diversidade de habitats físicos pode levar a uma simplificação das comunidades de organismos aquáticos (Busch & Lary, 1996).

Alterações na composição e abundância de organismos aquáticos devem ser analisadas de acordo com os seus desvios para as comunidades esperadas em áreas com condições naturais de referência, ou com um "ambiente com um bom potencial ecológico" no caso de reservatórios (Reynoldson et al., 1997; Reynoldson & Wright, 2000; Directive 2000/60/EC; Stoddard et al., 2006). Nos EUA, a United States Environmental Protection Agency (USEPA) elaborou um sistema de monitoramento utilizando um protocolo de avaliação de habitats físicos para lagos e rios (Kaufmann & Whittier, 1997; USEPA, 2007). O protocolo baseia-se em três elementos-chave para avaliação em ecossistemas lênticos: (i) avaliação das variáveis físicas e químicas da água; (ii) avaliação da região litorânea, bosque e sub-bosque ribeirinho em um transecto de 25 m de observação; (iii) descrição das características ripárias e do litoral em todas as estações de amostragem (USEPA, 2007). Levantamento de dados incluindo níveis de perturbações antrópicas, uso e ocupação do solo, aporte de nutrientes, tipo de substrato, entre outros, podem fornecer estimativas que, quando classificadas, ajudam a detectar estressores e a orientar programas de restauração de características físicas e químicas em reservatórios urbanos (Kaufmann & Whittier, 1997). Quantificar os descritores estruturais é uma maneira de garantir maior confiabilidade às análises (Kaufmann et al., 2008). Assim, modificações e adaptações do protocolo de diversidade de habitats desenvolvido para ecossistemas da América do Norte poderão contribuir como uma ferramenta integradora em programas de gestão de bacias hidrográficas brasileiras. O objetivo deste estudo foi avaliar a influência da complexidade estrutural dos habitats sobre as comunidades de macroinvertebrados bentônicos em reservatórios urbanos.

Material e Métodos

Foram amostrados três reservatórios urbanos na bacia hidrográfica do rio Paraopeba, afluente do rio São Francisco, Minas Gerais (Fig. 1).

O reservatório de Ibirité (20°01’13.39” S; 44º06’44.88” W) está localizado a uma altitude de 773 m. Apresenta acelerado processo de eutrofização artificial, o que se caracteriza por acentuada produção primária, com *blooms* de algas *Microcystis* sp. e de macrófitas aquáticas *Eichhornia crassipes* (Callisto et al., 2005; Moreno & Callisto, 2006). O reservatório de Vargem das Flores (19º 54’ 25.06’’ S; 44º09’17.78” W), localiza-se a 837 m de altitude e apresenta ocupação urbana em suas margens. O reservatório de Serra Azul (19º59’24.92” S; 44º20’46.74” W) localiza-se a uma altitude de 760 m, e juntamente com o reservatório de Vargem das Flores é considerado um importante manancial de abastecimento de água para a região metropolitana de Belo Horizonte (Morais et al., 2010).

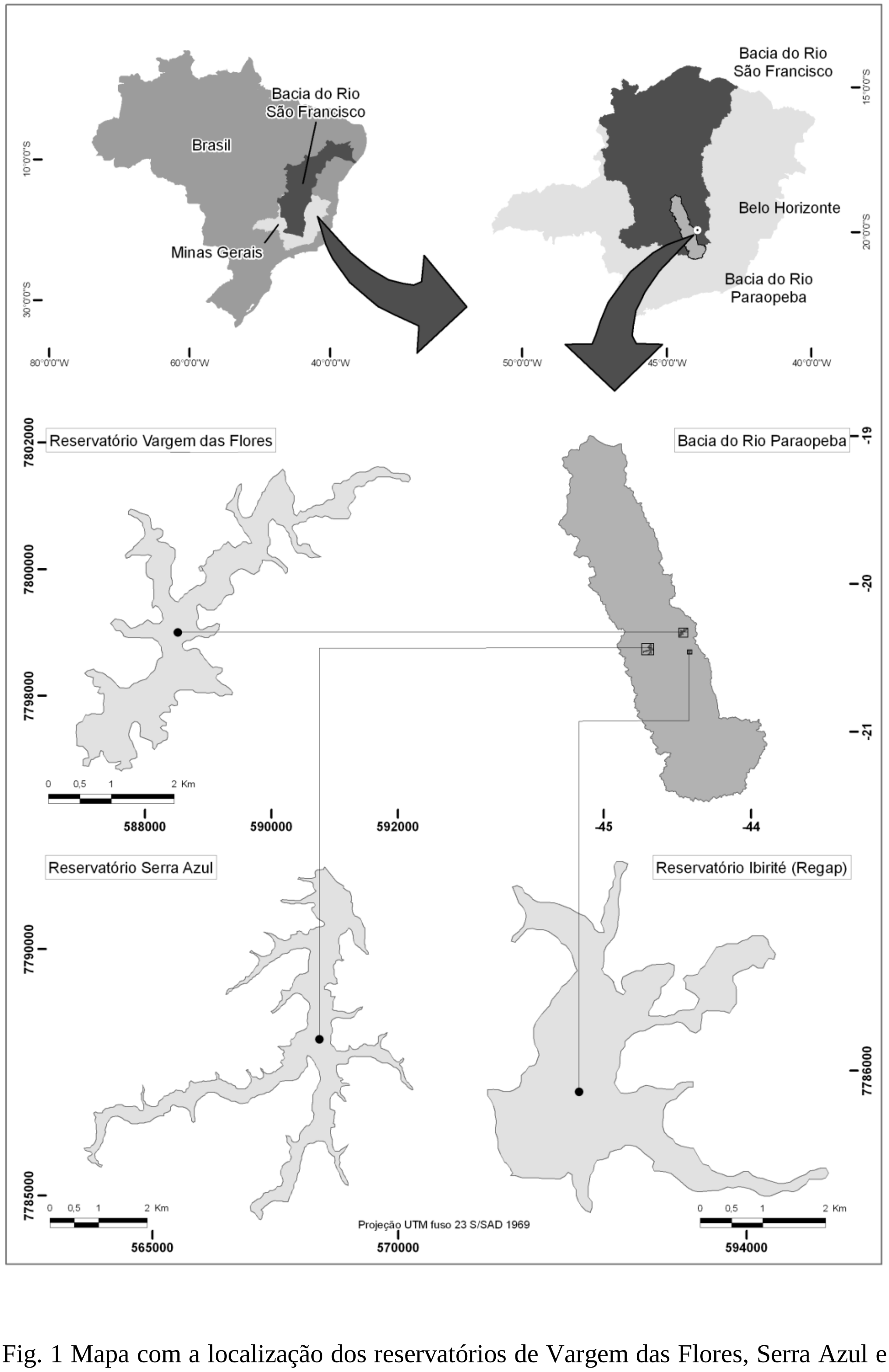

Fig. 1 Mapa com a localização dos reservatórios de Vargem das Flores, Serra Azul e Ibirité na bacia do Rio Paraopeba, Minas Gerais, Brasil.

Caracterização de habitats físicos

Foram realizadas avaliações semi-quantitativas nas zonas litorânea e ripária (bosque e sub-bosque) em 30 estações de amostragem numeradas de 1 a 30 em cada um dos três ecossistemas, totalizando 90 estações de amostragem. As coletas foram realizadas no período chuvoso de 2009 (dezembro), com nível de água mais elevado e menor distância entre a lâmina d'água e a zona ripária. A posição da primeira estação amostral nos três reservatórios foi próxima ao vertedouro e as demais foram selecionadas com o intuito de abranger todo o reservatório, distribuídas ao longo da região litorânea. Em cada uma dessas estações foi determinada uma área com medidas pré-determinadas (10 m de comprimento - região litorânea e zona de transição, estendendo-se mais 15 m para a zona ripária - bosque e sub-bosque por 15 m de largura), conforme o US Environmental Protection Agency (Baker et al., 1997; USEPA, 2007).

O protocolo de diversidade de habitats proposto por Baker et al. (1997) e USEPA (2007) foi adaptado e modificado para utilização em reservatórios urbanos. Este protocolo é composto por variáveis categóricas (representadas por códigos/letras) ou classificadas em faixas percentuais [(> 0-1%), 1 (>1-10%), 2 (>10-40%), 3 (>40-75%) e 4, (>75%)]. O USEPA (2007) recomenda que, para complementar a aplicação do protocolo de diversidade de habitats, sejam mensuradas variáveis físicas e químicas na água. A versão do protocolo traduzido e modificado para reservatórios urbanos encontra-se disponível no site do Laboratório de Ecologia de Bentos-UFMG (http://www.biogeral.icb.ufmg.br/benthos-Anexo I).

Variáveis físicas e químicas na coluna d’água

As coletas de água foram realizadas com garrafa do tipo Van Dorn na sub-superfície. As características físicas e químicas da água: pH, condutividade elétrica (µS cm^{-1}), turbidez (UNT), sólidos totais dissolvidos (mg L^{-1}), temperatura da água (°C) e oxigênio dissolvido (mg L^{-1}) foram mensuradas *in situ,* utilizando-se um multi-analisador modelo YSI. A leitura do disco de Secchi foi utilizada para avaliar a profundidade da zona eufótica (m) e a profundidade da coluna d’água foi estimada com aparelho portátil do tipo Sonar (m).

Amostras de água foram transportadas para o laboratório de Ecologia de Bentos na Universidade Federal de Minas Gerais em frascos de polietileno e resfriadas para determinação dos teores de Nitrogênio total, Fósforo total e Ortofosfato, de acordo com “Standard methods for the examination of water and wastewater” (American Public Health Agency, 2005). Para a análise das concentrações de clorofila-*a* na coluna d' água, amostras de 500 mL^{-1} foram filtradas, com posterior maceração dos filtros (Millipore AP47) e extração em acetona fria 90% conforme Golterman et al. (1978).

Composição granulométrica e teores de matéria orgânica nos sedimentos

A determinação da composição granulométrica dos sedimentos foi realizada segundo Suguio (1973), modificado por Callisto & Esteves (1996) pela metodologia de peneiramento, onde uma alíquota de 100 gramas foi seca em estufa (60^{o}C por 72 horas), e agitada em peneiras (16,00; 4,00; 2,00; 1,00; 0,50; 0,25; 0,125 e 0,062 mm). Os teores de matéria orgânica foram determinados pelo método de gravimetria pelo qual alíquotas (0,3 $\pm$ 0,1 g) foram calcinadas (550 ºC por 4 h), pesadas e a diferença entre o peso inicial da amostra e o peso após a calcinação forneceu a porcentagem do teor orgânico das amostras de sedimento.

Estrutura e composição de comunidades de macroinvertebrados bentônicos

Uma amostra de sedimento por estação de amostragem foi coletada com draga de Eckman-Birge (0,0225 m^2) nos mesmos pontos de coleta onde foi utilizado o protocolo de diversidade de habitats. O material coletado foi fixado *in situ* com formol tamponado, acondicionado em sacos plásticos e transportado até o laboratório, onde as amostras foram lavadas em peneiras com abertura de malhas de 1,00 e 0,50 mm (Larsen et al., 1991). As larvas de Chironomidae foram clareadas com solução de lactofenol 10% e identificadas com o auxílio de um microscópio (aumento de 400 x).

Análises estatísticas

As diferenças nos valores dos parâmetros físicos e químicos entre os reservatórios foram testadas através de análise de variância simples (ANOVA one-way) (Zar, 1999) com auxílio do programa STATISTICA 7.0. Para avaliar a estrutura das comunidades de macroinvertebrados bentônicos foram calculados o índice de diversidade de Shannon-Wiener, a densidade de organismos (indivíduos m^{-2}) e a dominância de ocorrência (% de indivíduos m^{-2}).

Uma análise de agrupamento (Cluster) foi realizada para verificar a similaridade entre as estações de observações nos diferentes reservatórios. Esta foi realizada no software PAST versão 2.04 (Hammer et al., 2001), utilizando o índice de distância de

Bray-Curtis e a UPGMA (Unweighted Pair Group Method with Arithmetic Mean) como método de amalgamação, sendo posteriormente avaliado o valor da correlação cofenética (Hammer et al., 2001).

Análises de correspondência canônica (CCA) foram realizadas com os dados transformados para log (x + 1). Estas análises foram realizadas a fim de avaliar as principais tendências de variações dos dados entre a composição e estrutura da fauna bentônica, composição granulométrica dos sedimentos, variáveis físicas e químicas da água e as variáveis do protocolo de diversidade de habitats físicos para cada reservatório. Para a identificação da importância relativa das variáveis foi utilizado o modelo *forward selection*, que descreve as variáveis estatisticamente significativas, com nível de significância α de 95% (p < 0,05). A significância das variáveis foi testada com 999 permutações. Das 68 variáveis mensuradas pelo protocolo de diversidade de habitats, foram eliminadas as que não apresentaram variações significativas e as correlacionadas de forma redundante com outras variáveis no conjunto de dados, utilizando o programa CANOCO for Windows 4.5 (Ter Braak & Smilauer, 1998).

Resultados

A aplicação do protocolo no reservatório de Ibirité evidenciou alterações de origem antrópica em 46,6% das estações, variando de moderadas a severamente. Nestes trechos a qualidade do ambiente está fortemente influenciada por atividades de pastagens, lixo, monocultura de *Eucalyptus*, residências, plantas industriais, lançamento de efluentes domésticos e industrial sem tratamento e controle de macrófitas aquáticas. Em Vargem das

Flores somente 34,8% das estações de amostragem apresentaram distúrbios relacionados a atividades de camping, residências, pastagens, mineração, desmatamento e pesca (tanques rede). Em Serra Azul 10% das estações apresentaram alterações antrópicas em pequeno grau, sendo que o único distúrbio foi a presença de residências.

Em Serra Azul 100% das estações apresentaram os maiores escores para as características “bem preservadas” e ambiente agradável. A zona ripária no entorno do reservatório é caracterizada por árvores grandes, e o sub-bosque é caracterizado por arbustos. No reservatório de Ibirité 100% das estações apresentaram alto grau de distúrbio e ambiente desagradável. Além disso, observa-se intenso deflorestamento da zona ripária onde o sub-bosque é caracterizado por ervas altas e gramíneas. Em Vargem das Flores, 50% das estações apresentaram os maiores escores para as características "bem preservadas", as demais estações foram caracterizadas como ambiente desagradável. Alguns trechos apresentam uma zona ripária com árvores grandes e um sub-bosque com arbustos, composto por plantas herbáceas e gramíneas.

As maiores variações na zona eufótica foram registradas nos reservatórios de Ibirité (0,27 ± 0,06 m) e Serra Azul (2,5 ± 0,63 m) (Tabela 1). Diferenças significativas foram encontradas entre os reservatórios de Serra Azul e e Ibirité, para as variáveis abióticas; condutividade elétrica (ANOVA $F_{2,58} = 21,26$; $p = 0,0001$), disco de Secchi (ANOVA $F_{2,69} = 14,80$; $p = 0,0001$), sólidos totais dissolvidos (ANOVA $F_{2,58} = 12,31$; $p = 0,005$), Nitrogênio total (ANOVA $F_{2,58} = 44,42$; $p = 0,0001$), Fósforo total (ANOVA $F_{2,58} = 12,72$; $p = 0,0001$) e clorofila-a (ANOVA $F_{2,58} = 12,04$; $p = 0,0002$). As variáveis turbidez (ANOVA $F_{2,58} = 21,39$; $p = 0,0001$) e Nitrogênio total (ANOVA $F_{2,69} = 2,12$; $p = 0,0001$) foram significativamente diferentes entre os reservatórios de Vargem das Flores e Ibirité.

Não foram observadas diferenças significativas entre os reservatórios de Serra Azul e Vargem das Flores.

Tabela 1. Variáveis abióticas (média e desvio padrão), composição granulométrica (%) e teores de matéria orgânica (% P.S.) mensurados no mês de dezembro de 2009 nos reservatórios de Serra Azul, Vargem das Flores e Ibirité, bacia do Rio Paraopeba, Minas Gerais, Brasil.

Variáveis	Reservatórios		
	Serra Azul	Vargem das Flores	Ibirité
Pedras % (64 - 250mm)	1,57 ± 6,70	0	0
Cascalho % (2 - 64mm)	1,36 ± 5,17	0,23 ± 0,35	0,25 ± 0,47
Areia Grossa % (1 - 0,50mm)	1,41 ± 2,45	14,30 ± 8,72	4,46 ± 6,70
Areia Média % (0,250 - 1mm)	15,68 ± 8,25	33,23 ± 16,30	22,15 ± 11,51
Areia Fina % (0,250 - 0,062mm)	19,09 ± 10,03	15,34 ± 11,18	25,26 ± 9,38
Silte/argila % (<0,062mm)	40,21 ± 18,46	17,98 ± 11,62	18,19 ± 17,36
Matéria Orgânica (% P.S.)	12,24 ± 3,19	8,17 ± 4,81	8,10 ± 5,03
Profundidade (m)	7,00 ± 4,08	3,89 ± 1,56	3,48 ± 2,92
Discos de Secchi (m)	2,50 ± 0,63	1,05 ± 0,28	0,27 ± 0,06
Temperatura do ar (ºC)	32,00 ± 0,45	28,20 ± 0,61	28,67 ± 1,53
Temperatura da água (ºC)	30,20 ± 0,44	27,96 ± 0,67	27,96 ± 0,67
pH	8,14 ± 0,44	8,48 ± 0,35	9,01 ± 0,50
Condutividade elétrica (µS cm^{-1})	26,13 ± 4,07	134,59 ± 13,60	285,81 ± 38,03
Sólidos totais dissolvidos (mg L^{-1})	16,77 ± 4,49	34,81 ± 5,53	177,35 ± 22,03
Oxigênio Dissolvido (mg L^{-1})	7,86 ± 0,28	7,56 ± 0,63	7,09 ± 2,15
Turbidez (NTU)	1,43 ± 0,53	4,84 ± 4,15	23,70 ± 4,22
Clorofila-*a* (µg L^{-1})	1,70 ± 2,09	3,00 ± 2,98	82,01 ± 122,72
Nitrogênio total (mg L^{-1})	0,06 ± 0,01	0,09 ± 0,04	0,26 ± 0,06
Fósforo total (µg L^{-1})	58,96 ± 25,53	26,65 ± 19,37	114,20 ± 50,94
Ortofosfato (µg L^{-1})	9,09 ± 8,84	6,93 ± 3,20	12,22 ± 21,58

No reservatório de Serra Azul houve predominância das frações areia média, areia fina e areia muito fina na maioria dos pontos amostrais (Tabela 1). Em Vargem das Flores a fração cascalho foi observada em proporções inferiores a 1%. No entanto, silte e argila representaram mais que 30% da composição granulométrica dos sedimentos. Não foram observados blocos de pedras no reservatório de Ibirité e a fração cascalho foi registrada em proporções inferiores a 2,3%. As frações de sedimento predominantes foram areia fina e areia muito fina, em 70% das estações. O sedimento é homogêneo em quase todas as estações e os teores de matéria orgânica não ultrapassaram 20% P.S., sendo que 55% das estações apresentaram valores inferiores a 10% P.S.

Foram coletados nos três reservatórios 831 organismos (Tabela 2). A riqueza de táxons foi maior no reservatório de Serra Azul (12 táxons) seguida por Vargem das Flores (11) e Ibirité (7). A densidade total de indivíduos foi maior no reservatório de Ibirité (84.400 ind m^{-2}), seguida de Vargem das Flores (27.173 ind m^{-2}) e Serra Azul (15.635 ind m^{-2}). Entre os grupos de macroinvertebrados, *Melanoides tuberculatus* Müller, 1774 representou 71,5% dos organismos amostrados, seguido por Chironomidae (19,1%).

No reservatório de Serra Azul foram coletados 116 indivíduos: 90,6% larvas de Diptera, sendo que destas 63,8% Chaoboridae. Entre os Chironomidae, representantes de Chironominae (14,7%) e Tanypodinae (12,1%) foram encontradas nas amostras, com destaque para *Coelotanypus* (12,1%), *Procladius* (5,2%), *Fissimentum* (2,6%), *Aedokritus*, *Alotanypus* e *Harnischia* (1,7% cada).

Em Vargem das Flores foram coletados 205 indivíduos: *Melanoides turberculatus* (34,2%), Oligochaeta (33,6%), *Corbicula fluminea* Müller, 1774 (5,9%) e Chironomidae (21,4%), com destaque para *Aedokritus* (6,3%) e *Coelotanypus* (2,4%). No reservatório de Ibirité foram coletados 510 indivíduos, 91,2% *Melanoides turberculatus*, 1,6% *Biomphalaria straminea* Dunker, 1848 e 6,7% Chironomidae, principalmente dos gêneros *Coelotanypus* e *Chironomus* representando 0,2% cada um (Tabela 2).

Tabela 2. Composição de macroinvertebrados bentônicos (média e desvio padrão) coletados no mês de dezembro de 2009 nos reservatórios de Serra Azul, Vargem das Flores e Ibirité, bacia do rio Paraopeba, Minas Gerais, Brasil.

Táxon	Reservatórios		
	Serra Azul	Vargem das Flores	Ibirité
Mollusca			
Gatropoda			
Prosobranchia			
Mesogastropoda			
Thiaridae			
Melanoides tuberculatus Müller, 1774	0,36 ± 0,59	2,50 ± 3,90	31,00 ± 55,64
Planorbiidae			
Biomphalaria straminea Dunker, 1848			0,53 ± 1,35
Bivalvia			
Corbiculidae			
Corbicula fluminea Müller, 1774		0,42 ± 0,99	0,06 ± 0,25
Annelida			
Hirudinea	0,05 ± 0,22	0,17 ± 0,54	
Oligochaeta		2,46 ± 4,61	0,13 ± 5,79
Insecta			
Ephemeroptera			
Polymirtacyidae		0,03 ± 0,18	
Baetidae		0,07 ± 0,37	
Leptoceridae		0,07 ± 0,37	
Odonata			
Gomphidae			
Diptera			
Chaoboridae			
Chaoborus Theobald, 1901	3,89 ± 7,52	0,89 ± 2,88	2,13 ± 5,79
Ceratopogonidae	0,15 ± 0,37		
Chironomidae			
Tanypodinae			
Coelotanypus Kieffer, 1913	0, 73 ± 1,24	0,17 ± 0,47	0,06 ± 0,25
Chironominae			
Aedokritus Roback, 1958	0,10 ± 0,31	0,46 ± 1,75	
Chironomus Meigen, 1803	0,05 ± 0,22		0,06 ± 0,25
Fissimentum Cranston & Nolte, 1996	0,10 ± 0,30		
Tanytarsus van der Wulp,1984		0,03 ± 0,18	
Harnischia Kieffer, 1921	0,10 ± 0,31		
Procladius Skuse, 1889	0,31 ± 0,82		
Alotanypus Roback, 1971	0,10 ± 0,31		
Parachironomus Lenz,1921	0,05 ± 0,22		

A análise de agrupamento apresentou valor de correlação cofenética de r = 0,89. Algumas estações apresentaram características de habitats semelhantes (estações 8, 11, 29, 9 e 28 em Serra Azul, e estações 19, 12 e 26 em Vargem das Flores) caracterizadas pela composição granulométrica dos sedimentos e por algumas características na região do entorno dos reservatórios. No entanto, não foi observado um padrão de diferenciação para os reservatórios na estrutura das comunidades, segundo a análise de Cluster (Fig. 2).

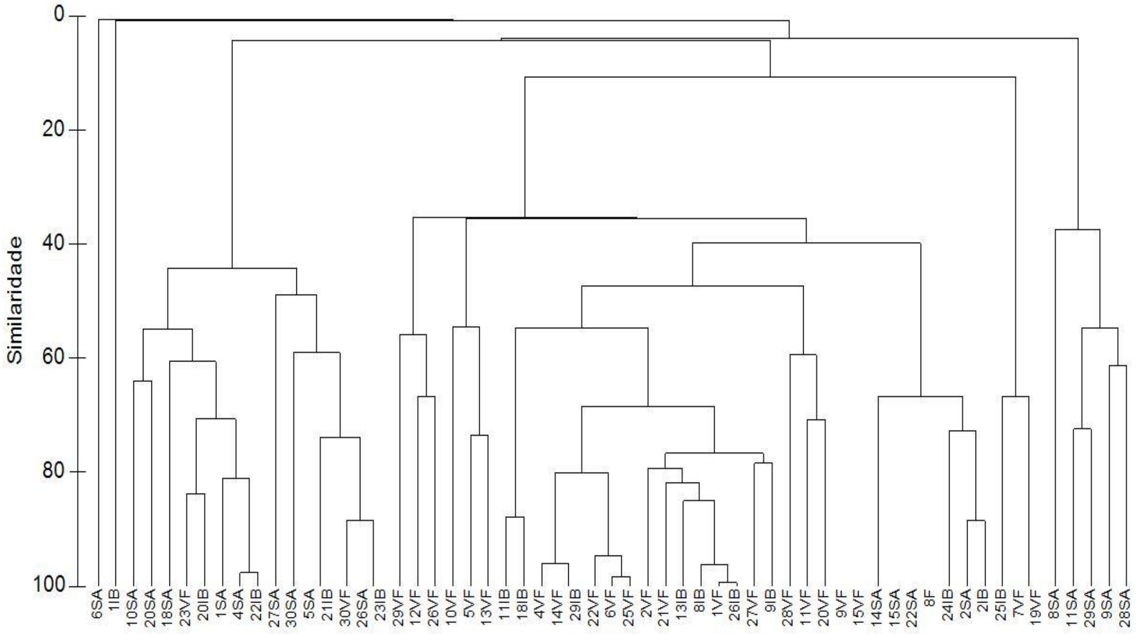

Fig. 2 Dendrograma de Similaridade da composição dos macroinvertebrados bentônicos nos reservatórios na bacia hidrográfica do rio Paraopeba, Minas Gerais, em dezembro de 2009. (SA, Serra Azul; VF, Vargem das Flores; IB, Ibirité).

Os eixos 1 e 2 da CCA explicaram 68,7% da relação entre as principais variáveis do protocolo de caracterização da diversidade de habitats e as comunidades bentônicas no reservatório de Serra Azul. O eixo 1 explicou 39,4% da variação e as variáveis relacionadas negativamente foram clorofila-*a*, profundidade e sub-bosque, enquanto que as variáveis associadas positivamente foram a fração granulométrica areia fina e o ângulo de inclinação. A fração areia fina explicou a relação entre o sedimento e as comunidades bentônicas de forma significativa (p = 0,049). Das 13 variáveis abióticas utilizadas pelo protocolo de diversidade de habitats, clorofila-*a* e profundidade foram significativas (p = 0,015 e p = 0,049, respectivamente) (Fig. 3A). As variáveis sub-bosque decidual herbácea e sub-bosque decidual arbusto, cobertura do solo, ervas altas, gramíneas e ângulo de inclinação do barranco foram correlacionadas, sendo cobertura do solo a única variável significativa (p = 0,035).

No reservatório de Vargem das Flores os eixos 1 e 2 explicaram 74,8% da relação entre as variáveis do protocolo e a comunidade bentônica. O eixo 1 explicou 41,6% da variabilidade. As variáveis clorofila-a, abundância total de macrófitas aquáticas, e macrófitas aquáticas emergentes estiveram positivamente relacionadas. Ortofosfato e fração areia grossa foram relacionadas negativamente. Areia grossa explicou de forma significativa a relação entre as frações de sedimento e a fauna bentônica (p = 0,049). Das variáveis abióticas, Ortofostato (p = 0,041) e clorofila-*a* (p = 0,036) foram significativas. Tipo de cobertura (p = 0,023), tipo do ambiente (p = 0,043), influência humana (comércio) (p = 0,022), influência humana (edifícios) (p = 0,013), zona de dossel (semi-decidual) (p = 0,015), abundância total de macrófitas aquáticas (p = 0,043), macrófitas aquáticas emergentes (p = 0,032), sub-bosque (com vegetação mista) (p = 0,049) e zona de transição (substrato composto por cascalho) (p = 0,043) foram significativas. No entanto, as únicas variáveis do protocolo que não apresentaram uma correlação redundante, e por isso foram utilizadas na CCA, foram abundância total de macrófitas aquáticas e macrófitas aquáticas emergentes (Fig. 3B).

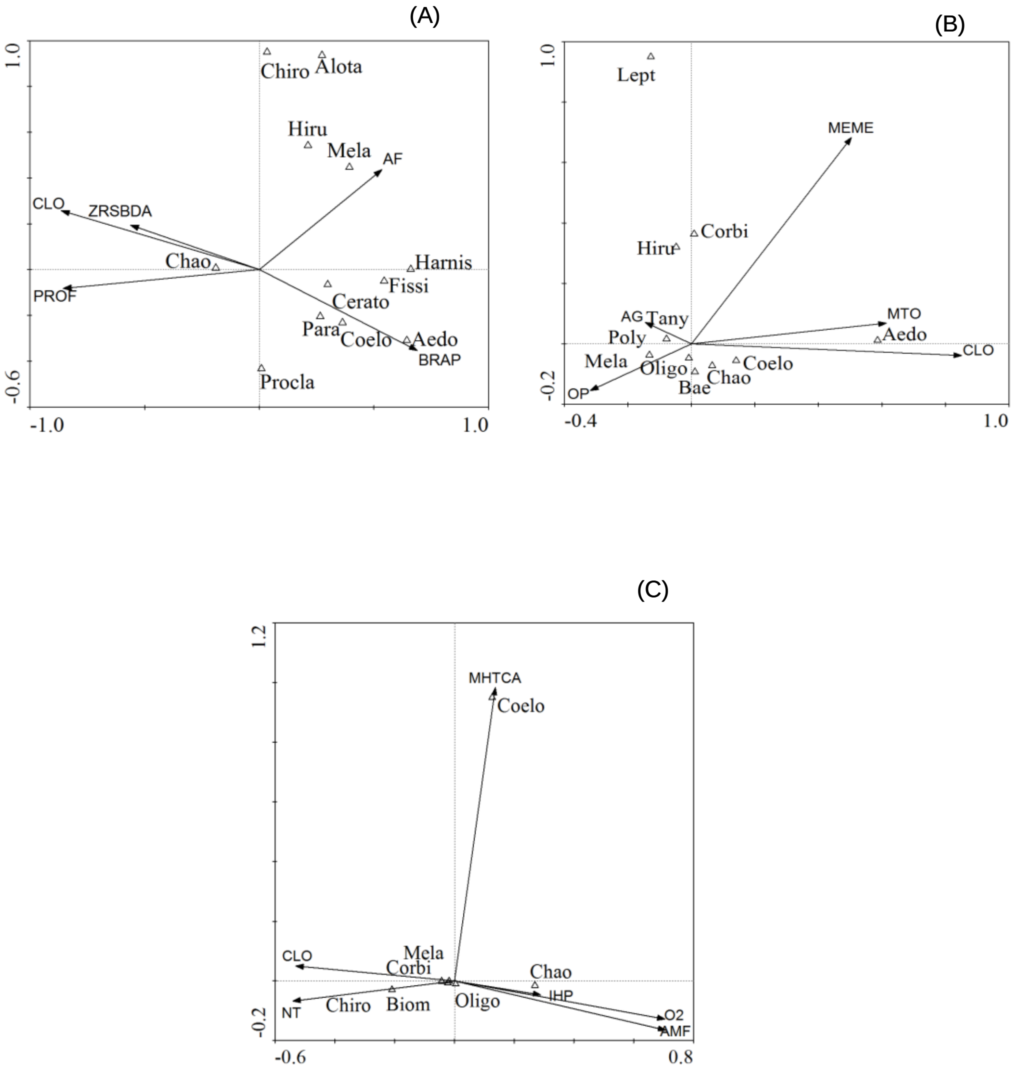

Fig. 3 Representação gráfica da Análise de Correspondência Canônica (CCA) entre composição do sedimento, variáveis físicas e químicas e variáveis de caracterização da diversidade de habitats relacionadas com os organismos bentônicos: A, Serra Azul; B, Vargem das Flores; C, Ibirité. AG, areia grossa (%); AF, areia fina (%); AMF, areia muito fina (%); IHP, influência humana (pastagem); MHTCA, tipo de cobertura (abundante); ZRSBDA, sub-bosque (decidual - arbustos); MTO, abundância total de macrófitas aquáticas; MEME, macrófitas aquáticas emergentes; BRAP, ângulo de inclinação do barranco (plano <5%); CLO, clorofila-*a* (µg L^{-1}); OP, orto-fosfato (µg L^{-1}); O_2, oxigênio dissolvido (mg L^{-1}); PROF, profundidade (m); NT, nitrogênio total (µg L^{-1}); Aedo, *Aedokritus*; Alota, *Alotanypus*; Bae, Baetidae; Biom, *Biomphalaria straminea*; Cerato, Ceratopogonidae; Coelo, *Coelotanypus;* Corbi, *Corbicula fluminea*; Chiro, *Chironomus;* Fissi, *Fissimentum*; Harnis, *Harnischia*; Hiru, Hirudinea; Lept, Leptoceridae; Mela, *Melanoides tuberculatus*; Oligo, Oligochaeta; Para, *Parachironomus*; Poly, Polymitarcyidae; Chao, Chaoboridae; Procla, *Procladius;* Tany, *Tanytarsus*.

Os eixos 1 e 2 da CCA explicaram respectivamente 34,6% e 18,6% da variância dos dados, totalizando 53,2% da explicação compartilhada entre as variáveis do protocolo de diversidade de habitats e a fauna bentônica no reservatório de Ibirité. As variáveis clorofila-*a* e Nitrogênio total relacionaram-se negativamente com o eixo 1, enquanto influência humana, tipo de cobertura, areia muito fina e oxigênio dissolvido relacionaram-se positivamente. A areia muito fina explicou de forma significativa a relação entre a fauna bentônica e a composição granulométrica dos sedimentos. Ortofosfato (p = 0,019), clorofila-*a* (p = 0,033), Nitrogênio total (p = 0,033), oxigênio dissolvido (p = 0,012), influência humana (pastagem) (p = 0,021), tipo de cobertura (vegetação) (p = 0,014), zona de dossel (ausência de vegetação) (p = 0,014), zona de transição (areia) (p = 0,033), distúrbios humanos (moderados) (p = 0,049) e macrófitas aquáticas (flutuantes) (p = 0,049) foram significativas. A variável do protocolo que não apresentou uma correlação redundante, e por isso foram utilizadas na CCA foi tipo de cobertura (Fig. 3C).

Discussão

No reservatório de Ibirité observa-se o processo de eutrofização artificial devido ao lançamento de esgotos domésticos sem tratamento (Callisto et al., 2005), elevados teores de nutrientes na coluna d’água (Moreno & Callisto, 2006), *blooms* de cianobactérias e macrófitas aquáticas flutuantes, baixos valores de visibilidade do disco de Secchi, provavelmente devido aos teores de matéria orgânica (Rodgher et al., 2005). O valor de Fósforo total na água foi superior ao mensurado nos reservatórios de Serra Azul e Vargem das Flores. No entanto, em algumas estações de amostragem foram registrados valores acima do limite da Resolução CONAMA 357/2005 (classe 2, 30 $\mu g\ L^{-1}$) (Brasil, 2005).

Os elevados valores de condutividade elétrica registrados nos reservatórios de Vargem das Flores e Ibirité provavelmente podem ser resultado do aporte de material carreado pelos rios a montante destes reservatórios. Lançamentos irregulares de efluentes industriais e/ou domésticos em bacias de drenagem refletem o nível de influência antrópica a que estes ecossistemas estão submetidos. Moreno & Callisto (2006), utilizando protocolo de caracterização ecológica rápida, verificaram avanço no processo de degradação na bacia hidrográfica do reservatório de Ibirité, com atividades agropastoris, lançamentos de esgotos não tratados e carreamento de sedimentos diretamente para o reservatório, o que persiste até os dias atuais.

A composição granulométrica de sedimentos tem sido considerada como um dos fatores que mais influencia a distribuição de macroinvertebrados bentônicos (Carvalho & Uieda, 2004). Esta abordagem baseia-se em observações recentes de que alguns táxons de macroinvertebrados bentônicos são restritos a tipos de substratos; e que tipos diferentes de substratos são capazes de hospedar comunidades de invertebrados que diferem em biomassa, densidade total e riqueza (Vitousek, 1990). Um substrato mais diversificado oferece maior disponibilidade de habitats e microhabitats (em uma escala de indivíduos), alimentos (diretamente ou adsorvidos nas partículas do sedimento) e proteção (por exemplo, de correntes e predadores, como peixes bentônicos) (Carvalho & Uieda, 2004; Maroneze et al., 2011). No presente estudo, a maior riqueza de organismos bentônicos foi encontrada no reservatório de Serra Azul, onde foi registrada maior diversidade granulométrica nos sedimentos e o mínimo de perturbações antrópicas na região litorânea.

O sedimento no reservatório de Ibirité é formado predominantemente por frações de areia fina e muito fina, caracterizando baixa heterogeneidade e disponibilidade de habitats físicos (Allan, 2004). Estudos em ecossistemas tropicais e temperados encontraram altas densidades de oligoquetos e baixa diversidade de outros macroinvertebrados associada a substratos formados por siltes e areias (Galdean et al., 2000; Fenoglio & Cucco, 2004). Assim, nossos resultados corroboram os estudos anteriores segundo os quais os oligoquetos foram encontrados associados a sedimentos formados por frações de areia e silte/argila ou areia fina, dominantes no reservatório de Vargem das Flores, onde predominaram sedimentos finos. A baixa riqueza de macroinvertebrados em sedimentos finos deve-se, provavelmente, a partículas próximas entre si e com menor conteúdo de água intersticial, reduzindo a captura de detritos de compostos orgânicos e a disponibilidade de oxigênio (Fenoglio & Cucco, 2004). Os oligoquetos de água doce vivem em todos os tipos de habitats, mas são mais abundantes em águas rasas, apesar de várias famílias terem representantes em lagos profundos (Alves et al., 2006).

Em Ibirité predominam *Melanoides tuberculatus* e *Biomphalaria straminea*. Estas são espécies exóticas e altamente competidoras que ameaçam a biodiversidade nestes ecossistemas (Souza & Lima, 1990; Silva et al., 2010). Indivíduos de *M. tuberculatus* são “r estrategistas” de reprodução partenogenética, com capacidade de manter altas densidades populacionais por longos períodos de tempo, além de possuírem grande capacidade migratória e fácil adaptação, estabelecendo-se em todos os tipos de substratos (Freitas et al., 1987). Por estas razões são encontrados em diversos ecossistemas aquáticos, com diferentes graus de trofia e poluição, incluindo ambientes oligotróficos a hipereutróficos (Freitas et al., 1987; Rocha-Miranda & Martins-Silva, 2006).

Os Chironomidae são os organismos mais representativos dos insetos aquáticos em decorrência da amplitude de ocupação de habitats; utilizam diversos recursos alimentares, o que confere estratégias adaptativas para colonizar diferentes tipos de micro-habitats por diferentes gêneros da família (Trivinho-Strixino & Sonada, 2006). Essa ampla faixa de ocorrência em termos de substratos para os Chironomidae pode ser explicada por seus hábitos alimentares diversificados (Armitage et al., 1994), podendo atuar como predadores, coletores de matéria orgânica particulada fina, e até eventualmente como fragmentadores (Tupinambás et al., 2007). De maneira geral, a família Chironomidae não apresenta exigências quanto ao substrato ideal para seu desenvolvimento (Entrekin et al., 2007).

Alguns gêneros registrados no reservatório de Vargem das Flores, como *Tanytarsus,* são habitantes de substratos arenosos sendo muito comuns em ambientes com baixo fluxo de água (Roque et al., 2004), sendo considerados indicadores de locais com boa qualidade de água (Panis et al., 1996). No reservatório de Serra Azul foram encontradas larvas de *Fissimentum* que, assim como *Tanytarsus,* são consideradas indicadoras de boa qualidade de água (Leal et al., 2004; Morais et al., 2010). Por outro lado, em Ibirité foram registradas altas densidades e biomassa de *Melanoides tuberculatus*, que ocorre em número elevado em ecossistemas aquáticos impactados por atividades antrópicas (Silva et al., 2010).

Quando avaliada a estrutura física dos habitats nos reservatórios, ficou evidente que algumas variáveis analisadas são extremamente importantes para a distribuição de comunidades bentônicas, tais como a influência humana, a presença de macrófitas aquáticas, o tipo do dossel e do sub-bosque da vegetação ripária. As alterações físicas decorrentes de atividades humanas (p. ex., deflorestamento da vegetação no entorno dos ecossistemas aquáticos provocadas por atividades de agricultura, pecuária e urbanização) podem causar maior degradação aos ecossistemas aquáticos do que processos de eutrofização ou de acidificação para as comunidades aquáticas (Busch & Lary, 1996).

As variáveis físicas mensuradas pelo protocolo evidenciam que algumas estações de amostragem podem apresentar potencial ecológico de referência nesta bacia hidrográfica, como algumas estações do reservatório de Serra Azul, que apresentam uma vegetação bem preservada, ausência de influência humana e um substrato diversificado (Reynoldson et al., 1997; Stoddard et al., 2006). As variáveis do protocolo de diversidade de habitats são medidas diretas ou indiretas do grau de conservação da diversidade e integridade ambiental (Sullivan, 2006). No reservatório de Ibirité foi encontrada menor riqueza e elevada abundância de *Melanoides tuberculatus*, se comparado aos demais. Além disso, este reservatório caracterizou-se por apresentar substrato homogêneo, intensa ocupação humana e atividades no entorno que provocam distúrbios na estrutura física dos habitats aquáticos.

Em conclusão, o grau de conservação do entorno dos reservatórios é um fator de extrema importância para habitats físicos e distribuição de comunidades bentônicas. Áreas com maior diversidade de habitats físicos oferecem abrigo e melhores condições para o estabelecimento de comunidades bentônicas. Nossos resultados evidenciam também a importância da utilização de um protocolo de avaliação da diversidade de habitats para relacionar com as comunidades de macroinvertebrados bentônicos em reservatórios urbanos. Assim, acreditamos que as informações obtidas com a utilização do protocolo de diversidade de habitats poderá ser uma ferramenta complementar no monitoramento de ecossistemas semi-lênticos, auxiliando os órgãos gestores na detecção de estressores e assim ajudando a orientar programas de restauração das características físicas e químicas em reservatórios urbanos no Brasil.

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ANEXO 1:

Quadro I - Protocolo de avaliação física de habitats proposto por Baker et al. (1997) e USEPA (2007), adaptado e modificado para ecossistemas semi-lênticos e lênticos.

CARACTERIZAÇÃO DE HABITATS FÍSICOS EM ECOSSISTEMAS SEMI-LÊNTICOS E LÊNTICOS

Local:____________________ Data:_____/_________/_____

Estação:_________________ Profundidade:________m GPS (UTM) :_______________/_______________

Zona litorânea							OBS
Biofilme na superfície	O Ausente		O Espuma				
	O Algas	O Óleo	O Outros				
Substrato de fundo							OBS
Rocha (>4000m)	0	1	2	3	4		
Matacão (250 - 4000mm)	0	1	2	3	4		
Bloco (Pedra) (64 - 250mm)	0	1	2	3	4		
Cascalho (2 - 64mm)	0	1	2	3	4		
Areia muito grossa (1,00 - 2,00mm)	0	1	2	3	4		
Areia grossa (0,50 - 1,00mm)	0	1	2	3	4		
Areia média (0,250 - 0,50mm)	0	1	2	3	4		
Areia fina (0,125 - 0,250mm)	0	1	2	3	4		
Areia muito fina (0,062 - 0,125mm)	0	1	2	3	4		
Silte + Argila (<0,062mm)	0	1	2	3	4		
Detritos foliares	0	1	2	3	4		
Matéria orgânica	0	1	2	3	4		
Cor do substrato	O Preto	O Cinza	O Marrom				
	O Verde		O Outro				
Cheiro	O Ausente	O H_2S	O Anoxia				
	O Óleo	O Químico	O Outro				
Macrófitas aquáticas							OBS
Submersas	0	1	2	3	4		
Emergentes	0	1	2	3	4		
Flutuantes	0	1	2	3	4		
Abundância de macrófitas	0	1	2	3	4		
Zona ripária							OBS
0 = Ausente (0%) 1 = Esparsa (<10%) 2 = Moderada (10-40%) 3 = Densa (40-75%) 4 = Muito densa(>75%)							
Dossel (>5m)							OBS
	O Decidual		O Semi-decidual				
	O Conífera	O Misto	O Ausente				
Árvores grandes (> 0,3 m - DAP)	0	1	2	3	4		
Árvores pequenas (< 0,3 m - DAP)	0	1	2	3	4		

Sub-bosque (0,5 a 5 m altura)							OBS
	O Decidual		O Semi-decidual				
	O Conífera	O Misto	O Ausente				
Arbusto	0	1	2	3	4		
Ervas altas e grama	0	1	2	3	4		
Cobertura do solo (0,5 m altura)							OBS
Paliteiros e mudas	0	1	2	3	4		
Plantas herbáceas e gramíneas	0	1	2	3	4		
Vegetação alagada	0	1	2	3	4		
Solo sem cobertura vegetal ou serrapilheira	0	1	2	3	4		
Substrato na área de inundação							OBS
Rocha (>4000mm)	0	1	2	3	4		
Matacão (250-4000mm)	0	1	2	3	4		
Bloco (Pedra) (64 - 250mm)	0	1	2	3	4		
Cascalho (2 - 64mm)	0	1	2	3	4		
Areia (0,062 - 2mm)	0	1	2	3	4		
Silte + Argila (<0,062mm)	0	1	2	3	4		
Detritos foliares	0	1	2	3	4		
Matéria orgânica	0	1	2	3	4		
Influência humana							OBS
Edifícios	0	1	2	3	4		
Comércio	0	1	2	3	4		
Estruturas humanas	0	1	2	3	4		
Estaleiros	0	1	2	3	4		
Muros e diques	0	1	2	3	4		
Aterros	0	1	2	3	4		
Estradas	0	1	2	3	4		
Linhas de transmissão	0	1	2	3	4		
Culturas	0	1	2	3	4		
Pastagens	0	1	2	3	4		
Agricultura	0	1	2	3	4		

Obs: Colocar F1, F2, F3 para observação

Quadro II - Protocolo de diversidade de habitat proposto por Baker et al. (1997) e USEPA (2007), adaptado e modificado para ecossistemas semi-lênticos e lênticos.

CARACTERIZAÇÃO DE HABITATS FÍSICOS EM ECOSSISTEMAS SEMI-LÊNTICOS E LÊNTICOS			
Local: ______________________		Data: _______/__________/______	
Classificação de macrohábitats para peixes na região litorânea		Características da margem (barranco)	
Distúrbio humano	O Ausente O Baixo O Moderado O Alto	Ângulo	O Plano (<5°) O Gradual (5 -30°) O Íngrime (30-75º) O Quase vertical (<75°)
Nível de Cobertura	O Sem cobertura O Cobertura esparsa O Cobertura contínua	Altura vertical do espelho flutuação da água	__________m
Tipo de cobertura	O Artificial O Pedregosa O Troncos O Vegetação O Nenhuma	Altura horizontal da água até a marca da flutuação	___________m
Substrato dominante	O Lama O Areia Grossa O Pedregoso O Rochoso		

Observações das atividades que caracterizam os distúrbios no ecossistema Intensidade: Não marcar = Não observado; B = Baixa, M = Moderada; E = Elevada				
Residencial	Recreação	Agricultura	Indústria	Ecossistema
B M E Residências	B M E Trilhas	B M E Plantações	B M E Plantas industriais	B M E Calagem
B M E Construções	B M E Parques	B M E Pastagens	B M E Minas	B M E Tratamento químico
B M E Diques	B M E Camping	B M E Pecuária	B M E Petróleo	B M E Pesca
B M E Entulho	B M E Resorts	B M E Pomares	B M E Usinas hidrelétricas	B M E Tratamento de água
B M E Estradas	B M E Marinas	B M E Animais confinados	B M E Desmatamento	B M E Controle macrófitas
B M E Pontes	B M E Lixo	B M E Captação de água	B M E Odores	B M E Flutuação da água
B M E Esgoto	B M E Espuma/Óleo	B M E Pecuária	B M E Comércio	B M E Tanques rede

Aparência geral do ecossistema			
Tipo hidrológico	O Reservatório	O Artificial	O Natural
Saída da barragem	O Ausente	O Artificial	O Natural
Linhas de transmissão (risco de vôo)	O Sim	O Não	
Densidade de barcos a motor	O Alta	O Baixa	O Proibída
Condições de nado	O Bons	O Aceitáveis	O Não permitido
Estruturas que indiquem alteração nível d'água	O Ausente	O Elevação = __________m	

Caracterização do Talude (%)				
Floresta	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Grama	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Arbusto	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Áreas alagadas	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Terra nua	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Agricultura	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Alteração nível d'água	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)
Residência	O Raro (<5%)	O Esparso (5-25%)	O Moderado (25-75%)	O Extenso (>75%)

Avaliação qualitativa de macrófitas aquáticas				
Emergentes/Flutuantes (% área do reservatório)	O < 5%	O 5 - 25%	O 25 - 75%	O > 75%
Submersas (% área do reservatório)	O < 5%	O 5 - 25%	O 25 - 75%	O > 75%
Abundância de macrófitas (%)	O Ausente	O Esparsa	O Moderada	O Alta

Caracterização do corpo d´água						
"Pristine"	O 5	O 4	O 3	O 2	O1	Alto Distúrbio
Agradável/prazeroso	O 5	O 4	O 3	O 2	O1	Desagradável/não prazeroso

OBS:

CAPÍTULO 4

Maximum ecological potential of tropical reservoirs and benthic macroinvertebrate communities

Joseline Molozzi[1], Maria João Feio[2], Fuensanta Salas[3],

João Carlos Marques[2] & Marcos Callisto[1]

Reservatório de Serra Azul

1- Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, ICB, Universidade Federal de Minas Gerais, Brasil
2- Intituto do MAR-CMA, Departamento de Ciências da Vida, Universidade de Coimbra, Portugal
3- TRAGSA, Murcia, Espanha

Submetido: Hydrobiologia

Maximum ecological potential of tropical reservoirs and benthic macroinvertebrate communities

Joseline Molozzi, Maria João Feio, Fuensanta Salas,

João Carlos Marques & Marcos Callisto

Abstract: The Reference Condition Approach (RCA) is currently universally adopted as a basis for the evaluation of the ecological quality of water bodies. The RCA was also used here with the aim of defining the Maximum Ecological Potential (MEP) of tropical reservoirs located on the hydrographic basin of the Paraopeba River, Minas Gerais State – Brazil. Ninety sites located in three reservoirs were analysed and sampled every three months over 2 years for benthic macroinvertebrate communities. The communities' temporal patterns were previously analysed (2nd STAGE-MDS and ANOSIM) in the three reservoirs and were not significantly related to the seasonal fluctuations in temperature and precipitation. Twenty-eight sites with the lowest human disturbance levels were selected to define the MEP. All these sites were located in Serra Azul, a reservoir used for water supply and contained in a special protected area, where tourism is not allowed and the native vegetation is conserved. The macroinvertebrate taxa present in the MEP selected sites are similar to those of natural lakes of the region and different from the communities of disturbed sites. The biological classification of these sites showed two groups with distinct macroinvertebrate communities. This distinction was related to bottom and shoreline substrate and depth. These two subsets of biological communities and respective environmental conditions are a basis for future development of a quantitative assessment system to monitor tropical reservoirs in the study area.

Keywords: Reservoirs, macroinvertebrates, reference condition approach, tropical region.

Introduction

The ecological quality of an aquatic ecosystem is defined nowadays in a holistic manner, combining information acquired from the traditional monitoring of physical and chemical parameters with that of the biological communities. In accordance with the Reference Condition Approach (Reynoldson et al., 1997; Reynoldson et al., 2001; Bailey et al., 2004; Stoddard et al., 2006), which is currently used worldwide and is the basis of the European Water Framework Directive (Directive, 2000/60/EC), the integrity of communities found in one location should be analysed according to their deviation to expected communities in the absence of anthropogenic disturbances (Nijboer et al., 2004; Ruse, 2010; Hawkins et al., 2010). Thus, it is essential to know how the communities of a given ecosystem would be like in the absence of impacts.

Nonetheless, in practice, an ecosystem in reference condition rarely corresponds to the concept of pristine, since the most commonly used method to define reference is based on recently sampled local communities, and for most regions the total absence of anthropogenic impacts does not exist (Gibson et al., 2000; Stoddard et al., 2006). Moreover, in the case of reservoirs, the concept of pristine cannot be used at all, since these are heavily modified water bodies where the environment shifted from lotic to lentic (Nilsson et al., 2005) and significant changes in the structure of rivers and hydrographical basins and in their hydrological regimes occurred (Tundisi & Matsumura-Tundisi, 2003).

In Europe, and according to the European Water Framework Directive (Directive, 2000/60/EC; WFD, 2003), the term Maximum Ecological Potential (MEP) is used to define the best status that a heavily modified or artificial water body can achieve. The MEP status may include permanent hydromorphogical changes but after all mitigation measures have been considered and assuming a suitable water quality (Irmer & Pollard, 2006, Lammens et al., 2008).

The use of an entire reservoir as reference or alternatively the choice of individual sites, as done in rivers, is not consensual. Navarro et al. (2009) consider that, due to the difficulty in finding unpolluted reservoirs, the use of a reservoir presenting good ecological quality as reference for other reservoirs with similar abiotic characteristics is acceptable. However, we defend here, as other authors (Gibson et al., 2000; Dodds et al., 2006), that the entire reservoir should not be used as reference since that approach does not represent the diversity of the physical, chemical or even biological characteristics of the whole. Moreover, within a given reservoir there are heavily impacted regions and others not so impacted depending on the human activities and proximity to urban centres (Kennedy, 2001; Yanling et al., 2009).

There are essentially two ways to identify reference sites for assessment purposes: a priori classification (typology), based on the abiotic characteristics of the sites (e.g. altitude, drainage area, latitude, longitude), which the Water Framework Directive is consistent with (Directive, 2000/60/EC; Piet et al., 2004; Salas et al., 2006; Teixeira et al., 2007; Puntí et al., 2007); and a posteriori classification, used by the majority of predictive models (e.g. RIVPACS/AUSRIVAS, BEAST; Reynoldson et al., 1997; Clarke et al., 2003; Bailey et al., 2006; Feio et al., 2007, 2010; Aroviita et al., 2010). In a comparative study, Davy-Bowker et al. (2006) concluded that the first approach depends heavily on how well the variables used in the formation of types correlate with the ecological characteristics of the communities. In the scope of this study, and given that reservoirs are less studied systems than rivers, a posteriori classification system was considered more adequate since it provides the determination of which environmental variables best explain the distribution of fauna in the different groups of sampling stations.

Benthic macroinvertebrates are widely used for the bioassessment of rivers and streams since they are relatively easy to sample and identify, and they reflect the surrounding environmental conditions (Bonada et al., 2006). They are also among the biological elements recommended by the WFD for the evaluation of reservoirs ecological quality status even though in practice they are not commonly used due to practical difficulties in sampling methods. Instead fishes and phytoplankton are the most frequent biological elements evaluated in reservoirs (e.g. Navarro et al., 2009).

Therefore, the aim of the present study is to define Maximum Ecological Potential (MEP) of tropical reservoirs of Minas Gerais, Brazil, based on the selection of the best available sites based abiotic pressure data (hydromorphological measures and water physical-chemical measurements) and the characterization of their benthic macroinvertebrate communities. We expect to find most of the best sites within the reservoir of Serra Azul, which is included in a special protected area, with dense native vegetation, limited human access, and where no tourism or fisheries are allowed. Furthermore, we investigated the influence of seasonal variations in the macroinvertebrate communities of the reservoirs, to determine the need of defining different Maximum Ecological Potential values for different seasons. This would be the basis for the development of a future assessment tool, which will help to control and improve the ecological status of present and planed reservoirs of this tropical.

Methods

Study area

In total, 90 sampling sites were sampled in the littoral region of 3 reservoirs (Ibirité, Vargem das Flores and Serra Azul) in the Paraopeba river watershed, an affluent of the São Francisco River in the Minas Gerais state, south-eastern Brazil. The climate of this region is considered tropical sub-humid (Cwb), with summer rains (November to April) and a dry winter (May to October). The average annual temperature is ca. 20^{o}C (Moreno & Callisto, 2006) (Fig. 1).

The Ibirité reservoir (20º 01’ 13.39” S; 44° 06’ 44.88” W) was built in 1968 at an altitude of 773 m a.s.l. This reservoir has an area of 2.8 km^2, a volume of 15,423.000 m^3 and an average depth of 16 m. The hydrographic basin of the Ibirité Reservoir extends over two municipalities, Ibirité (148,535 inhabitants) and Sarzedo (23,282 inhabitants). The landscape of the reservoir basin is dominated by Eucalyptus plantations, a large condominium, small farms, and several industrial plants (Pinto-Coelho et al., 2010).

The Vargem das Flores reservoir (19º54’25.06’’ S; 44º 09’ 17.78) was built in 1971 and is situated at 837 m a.s.l as is for water supply for the cities of Contagem and Belo Horizonte. The reservoir has a surface area of 4.9 km^2, contains 37,000.000 m^3 of water and has a maximum depth of 18 m. The maximum height of sill spillway is 838,64 m and the reservoir as a hydraulic retention time of 365 days (HDC, 2000; COPASA, 2004). About 12.3 ha of the area around the reservoir were transformed into a special protected area of the state of Minas Gerais in 2006 (COPASA, 1980a). However, the population around the reservoir reaches about 100.000 people, and was transformed into area leisure for the region. The Serra Azul reservoir (19° 59’ 24.92” S; 44° 20’ 46.74” W), located at an altitude of 760 m a.s.l., has a water surface of 7.5 km^2, water volume of 88,000.000 m^3 and a maximum depth of 40 m. It has been operating for approx, 30 years as a source of drinking water to the metropolitan region of the State’s capital (ca. 4.8 million people). The maximum height of sill spillway is 760 m and the reservoir has a hydraulic retention time of 351 days. Surrounding this reservoir there is also a special protected area established in 1980 with an area of 27.000 ha. Inside this area, 3.2 ha belong to the COPASA (1980b), the industry that explores the reservoir and no tourism or fisheries are allowed. The landscape is mostly covered with native vegetation and an effort was made in order to remove exotic plants covering spot areas and substitute it by autochthonous vegetation. The only human pressure existent in the area is due to a small number of houses (≈20) remaining in a constrained area, from the period of construction of the reservoir.

Climatic data

To analyse the seasonal patterns, the average monthly values of temperature and precipitation were calculated for all sampling periods based on data from the Brazilian National Institute of Meteorology (INMET) for the metropolitan region of Belo Horizonte in 2008 and 2009.

Environmental data

With the purpose of characterizing the natural conditions found in the reservoirs and also to distinguish the various sites in respect to their level of anthropogenic disturbance several parameters related to the water chemistry and physics, to hydromorfology and to land use were measured in situ and are summarized in Table 1.

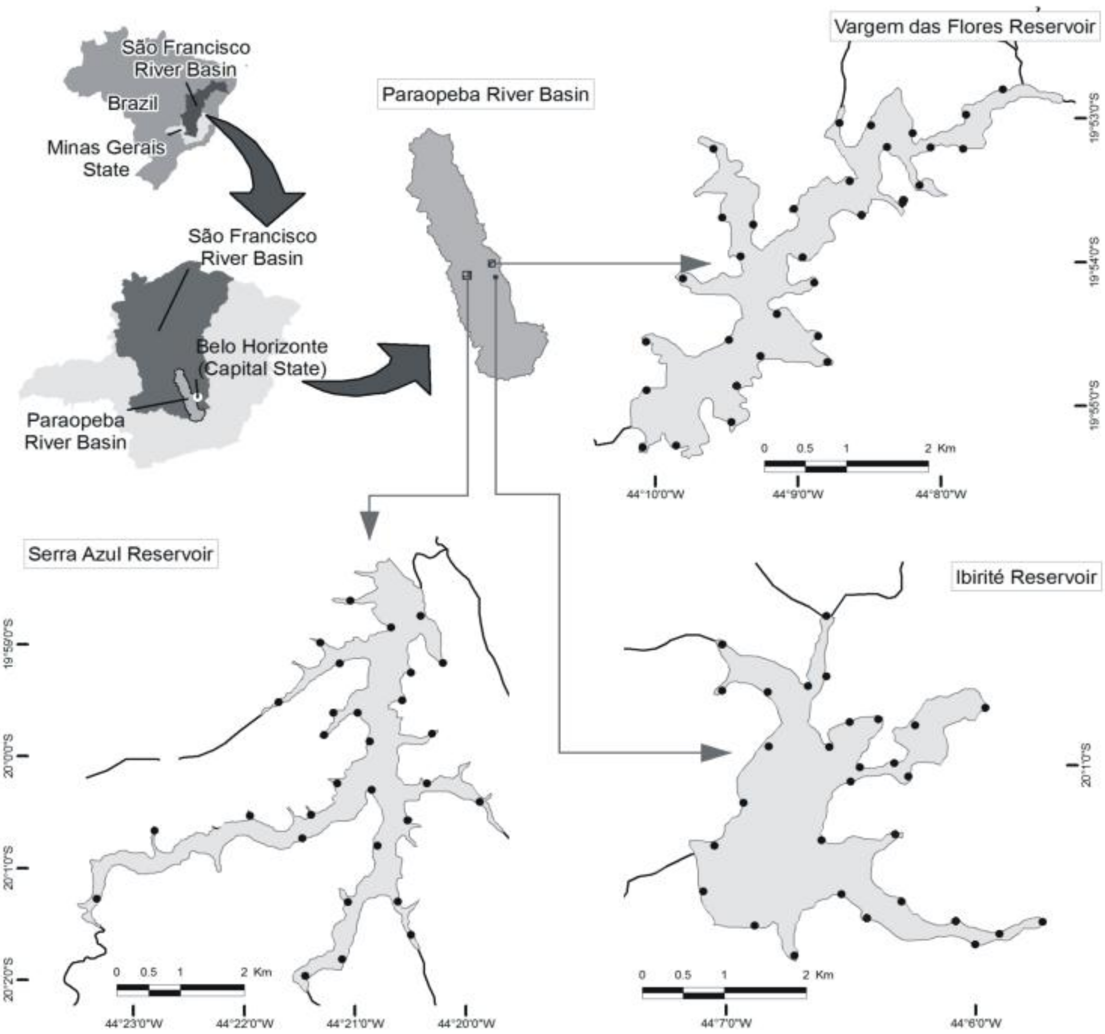

Fig. 1 Location of the reservoirs of Vargem das Flores, Serra Azul and Ibirité in the catchment of the Paraopeba River, Minas Gerais, Brazil and distribution of the sampling sites (black dots) in the reservoirs.

At each sampling occasion the following physical and chemical parameters of water were measured using an YSI Model-Multiprobe: water temperature, dissolved oxygen, conductivity, turbidity and pH (Table 1). Additionally, sub-surface water samples were collected with a Van Dorn type cylinder for subsequent measurement of total nitrogen (TN), total phosphorus (TP) and orthophosphates (PO_4), in accordance with "Standard Methods for the Examination of Water and Wastewater " (APHA, 1992). The concentration of chlorophyll *a* (Chla) was obtained according to Golterman et al.

(1978). The depth of the eufothic zone (S) was determined based on the readings of the Secchi disc.

The Carlson (1977) trophic state index (TSI1), modified by Toledo et al. (1983), and the Trophic State Index proposed by the Brazilian Society of Enviromental Agency Technology (CETESB, 2000) (TSI2), were calculated for all sites. Each index is composed sub-indices, which are then weighted to obtain a final value for the trophic status. The TSI1 is calculated through the formula

$$TSI1 = TSI (S) + 2 *[TSI (TP) + TSI (P0_4) + TSI (Chla)/7]$$

and the sub-indices are obtained as follows:

$$TSI (S) = TSI (S) = 10 * (6 - (0.64 + \ln S) / \ln 2))$$

$$TSI (TP) =10 * (6 – (\ln (80.32 / TP) / \ln 2))$$

$$TSI (PO_4) = 10 * (6 – (\ln (21.67 / PO4) / \ln 2))$$

$$TSI (Chla) =10 * (6 – (2.04 – 0.695 \ln Chla) / \ln 2))$$

The TSI2 is calculated through the formula

$$TSI2= [TSI (TP) + TSI (Chla)]/2$$

and the sub-indices are obtained through the expressions:

$$TSI (TP) =10 * (6 – ((1.77-0.42)* \ln (TP) / \ln 2))$$

$$TSI (Chla) =10 * (6 – (0.92 – 0.34)* \ln Chla) / \ln 2))$$

TSI1 values ranging from 0 to 44 correspond to oligotrophic, 44–54 to mesotrophic, and > 54 to eutrophic waters. TSI2 values ranging from 0 to 23 correspond to ultraoligotrophic, 24–44 to oligotrophic, 44–54 to mesotrophic, 54–74 to eutrophic, and > 74 hypereutrophic conditions.

Table 1. Description of the environmental variables measured at all sites.

Environmental Variables	**Description and Source**	**Mean (Range)**
Pressure variables		
Total dissolved solids (TDS; mg L^{-1})	Field measurement (YSI)	106.07 (9.04 - 324.45)
Chlorophyll *a* (Chla; µg L^{-1})	Analysis according to Golterman et al. (1978)	19.35 (0 - 228.04)
Total nitrogen (TN; mg L^{-1})	Analysis according to APHA (1992)	0.19 (0.01 - 1.37)
Total phosphorus (TP; µg L^{-1})	Analysis according to APHA (1992)	72.14 (2.35 - 789.35)
Orthophosphates (PO_4; µg L^{-1})	Analysis according to APHA (1992)	2.60 (2.03 - 284.41)
Odour of bottom substrate	Field observation. Categories: 1(none), 2(H2S), 3(anoxic), 4(oil), 5 (chemical), 6(other) - USEPA (2007)	0 - 1
TSI1	Analysis based on Carlson (1977), modified by Toledo et al. (1983)	46.62 (34.90 - 84.21)
TSI2	Analysis based on CETESB (2000)	62.00 (25.46 - 91.00)
Buildings (%)	Field observation. Categories: 1=absent (0%), 2=sparse (10%), 3=moderate (10 - 40%), 4=heavy (40 - 75%), 5=very heavy (>75%). USEPA (2007)	1 - 2
Commercial buildings (%)	Idem	1 - 3
Docks/boats (%)	Idem	1 - 4
Dykes (%)	Idem	1 - 3
Landfills (%)	Idem	1 - 2
Roads (%)	Idem	1 - 3
Power lines (%)	Idem	1 - 3
Row crops (%)	Idem	1 - 3
Pasture (%)	Idem	1 - 3
Agriculture (%)	Idem	1 - 2
Characterization variables		
Gravel/ boulders – bottom (4m - 2mm)	Field observation. Categories: 1= 0%, 2= >0 - 20%, 3= >20 - 60%, 4= >60%	1.19 (0.24 - 6.47)
Coarse sand – bottom (2 - 0.50mm)	Field observation. Categories: 1= 0 - 15%, 2= >15 - 35%, 3= >35 - 45%, 4= >45%	16.62 (0 - 51.74)
Fine sand – bottom (0.50 - 0.062mm)	Field observation. Categories: 1= 0 - 20%, 2= >20 - 50%, 3= >50 - 80%, 4= >80%	42.52 (0 - 92)
Silt, clay or muck – bottom (<0.062mm)	Field observation. Categories: 1= 0 - 15%, 2= >15 - 35%, 3= >35 - 45%, 4= >45%	25.47 (0 - 85.50)
Bedrock – shoreline (>4m)	Field observation. Categories: 1= 0 - 15%, 2= >15 - 35%, 3= >35 - 45%, 4= >45%	1 - 2
Cobble – shoreline (64 - 4000mm)	Field observation.Categories: 1= 0 - 15%, 2= >15 - 35%, 3= <35 - 45%, 4= >45%	1- 2
Gravel – shoreline (2 - 64mm)	Field observation. Categories: 1= 0 - 15%. 2= >15 - 35%. 3= >35 - 45%. 4= >45%	1 - 2
Sand/muck – shoreline (0.062 - 2mm)	Field observation. Categories: 1= 0 - 15%, 2= >15 - 35%, 3= >35 - 45%, 4= >45%	1 - 4
Depth (m)	Field measurement (sonar)	3.92 (0.4 - 16.20)
Bank steepness	Field observation. Categories:1=flat (<5º), 2=gradual (>5 - 30º), 3=steep (>30 - 75º), 4=near vertical (>75º)	1- 4

To characterize the littoral, transition and riparian zones near the sampling sites, we followed the protocol for lentic ecosystems proposed by EMAP-USEPA

(Environmental Protection Agency, EUA; USEPA, 2007). Data was recorded in December 2009, at each site. in a transect with an area of 15 m width x 25 m long. In this 25m are included 10 m in the littoral and transition zones and 15 m in the riparian zone. The variables included in the protocol and used in this study are described in Table 1 and are related to land use, type of sediment and depth. The depth of the water column was estimated using a portable sonar. Sediment collected with the Eckman-Birge dredge was analyzed regarding its granulometric composition and organic matter content, according to the Suguio (1973) methodology, modified by Callisto & Esteves (1996).

Macroinvertebrate sampling

The reservoirs were sampled in 90 sites quarterly (March, June, September and December) in 2008 and 2009, with an Eckman-Birge dredge (0.0225 m^2), as close as possible to the margin and at a depth varying from 0.4 to 16.2 m (mean depth of 3.92 m). The collected material was fixed with 70% formalin and was transported to the laboratory. Invertebrates were mostly identified to the family level (Peterson, 1960; Pérez, 1988; Merritt & Cummins, 1996; Carvalho & Calil, 2000; Fernandez & Dominguez, 2001; Costa et al., 2006; Mugnai et al., 2010). Chironomidae larvae were identified to genus, treated with 10% solution of lactophenol and identified under a microscope (400x) with the aid of Trivinho-Strixino & Strixino (1995) and Epler (2001) identification keys.

Data analyses

Determination of biological seasonal variability

The similarity between communities in different seasons and years was analysed for each reservoir with a 2nd STAGE non-metric Multidimensional Scaling Analysis (nMDS) (Clarke & Gorley, 2006). This MDS is based on the similarity matrix resulting from a 2nd STAGE analysis. This procedure calculates a similarity matrix based on the Spearman rank correlation between pairs of Bray-Curtis similarity matrices, each one composed of the biological data collected at a given season and year.

Additionally, an Analysis of Similarity (ANOSIM) was done to test whether the benthic communities were statistically similar between seasons, for the two-year sampling period.

Selection of sites with Maximum Ecological Potential

A Principal Components Analysis (PCA) on pressure data described in Table 1 (normalized data; Clarke & Warwick, 2001) was carried out for all sites and samples in order to determine which are the sites least affected by human disturbance and therefore used to define the Maximum Ecological Potential and which are the most relevant pressures in the study area. Additionally, the distribution of values of each pressure variable was inspected with box plots and the sites with outliers values were subsequently removed from the MEP data set. For the final set of MEP sites, the range, (ninimum and maximum) of pressure variables were calculated in order to define intervals of values of acceptable pressures for these systems.

To verify whether the biological communities of the selected MEP sites were, in general, distinct from those affected by a higher level of pressure, we performed a non-metric Multidimensional Scaling (nMDS) ordination with the biological data (square root transformation; Bray-Curtis similarity) (Clarke & Warwick, 2001).

Establishment of subsets of communities in Maximum Ecological Potential

UPGMA (Unweighted Pair Group Method with Arithmetic Mean; Bray-Curtis similarity; square root transformation) cluster analysis was carried out to analyse whether there are sub-sets of reference conditions (groups of sites with similar communities) in the selected MEP sites. The statistical difference among the groups was tested by ANOSIM.

In order to determine the most representative species of each group and to verify if they differed among the reference groups, the SIMPER analysis was used (Clarke & Warwick, 2001). The total number of individuals, number of species, Margalef's richness (Margalef, 1969), Shannon-Wiener's diversity (Shannon & Weaver, 1963), and Pielou's evenness (Pielou, 1969) further characterized the different groups found.

Abiotic typology

A stepwise forward discriminant analysis (Alpha-to-Tolerance = 0.001 and Alpha-to-Remove = 0.10 with Jackknife cross-validation, Hair et al., 1998) was performed to find the environmental variables that best distinguish the communities in the groups. The potential discriminating variables used in the analysis (Table 1) describe the morphological characteristics of the system and were selected as being the variables that are less subject to changes of anthropogenic origin, such as the type of substrate and the slope of the shoreline.

All statistical analyses were performed using the PRIMER 6.0 software, excepting the Discriminant Analysis, which was performed using Systat 13.0 (Systat Software, Cranes Software International Ltd. 2008).

Results

Seasonal variability

In total, 14,425 organisms, belonging to 47 taxa (4 Mollusca, 2 Annelida and 41 Arthropoda), were collected in 90 sampling sites over two years. Of the total number of organisms sampled, 24% were Diptera larvae, where *Chironomus* (8%), *Tanypus* (4%) and *Coelotanypus* (4%) were the most representative genera (Appendix I).

The climatic data for the years 2008 and 2009 confirmed the existence of a distinct wet season (December and March), and a dry season (June and September), December 2008 was the month with the highest average rainfall (442 mm), followed by January 2009 (282 mm). The driest periods were the months of June 2008 and 2009, with no precipitation. The maximum temperatures during the study period were recorded in December 2009 (29.0 ºC) and the minimum temperatures in June 2008 (23.9 ºC) (Fig. 2).

The 2nd STAGE - nMDS was not consistent with the above, showing that there is no pattern of high correlation between the communities sampled in the same month of the year (e.g. December 2008, December 2009) or the same season (dry, wet) (Fig. 3). The R values of the global ANOSIM for the three reservoirs showed a wide variability within the sampling periods (ANOSIM Serra Azul: Global R = 0.054, p = 0.001; ANOSIM Ibirité: R = 0.166, p = 0.001; ANOSIM Vargem das Flores: R = 0. 113, p = 0.001) which was confirmed by most pairwise tests (Table 2).

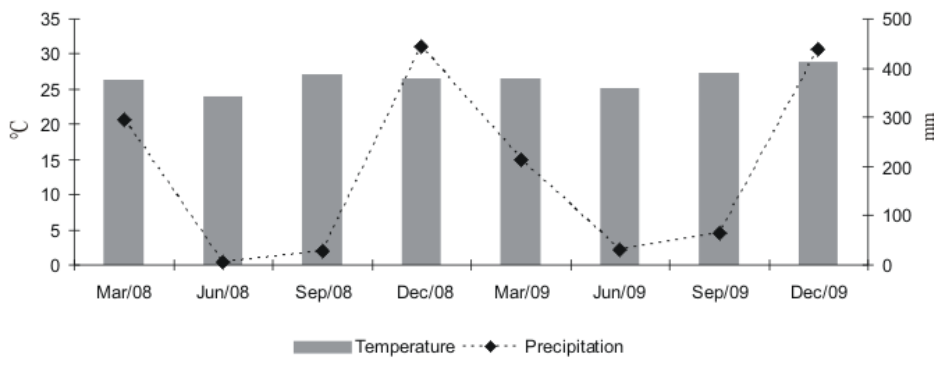

Fig. 2 Average monthly precipitation (mm; bars) and temperature (ºC. dots) observed during the sampling periods.

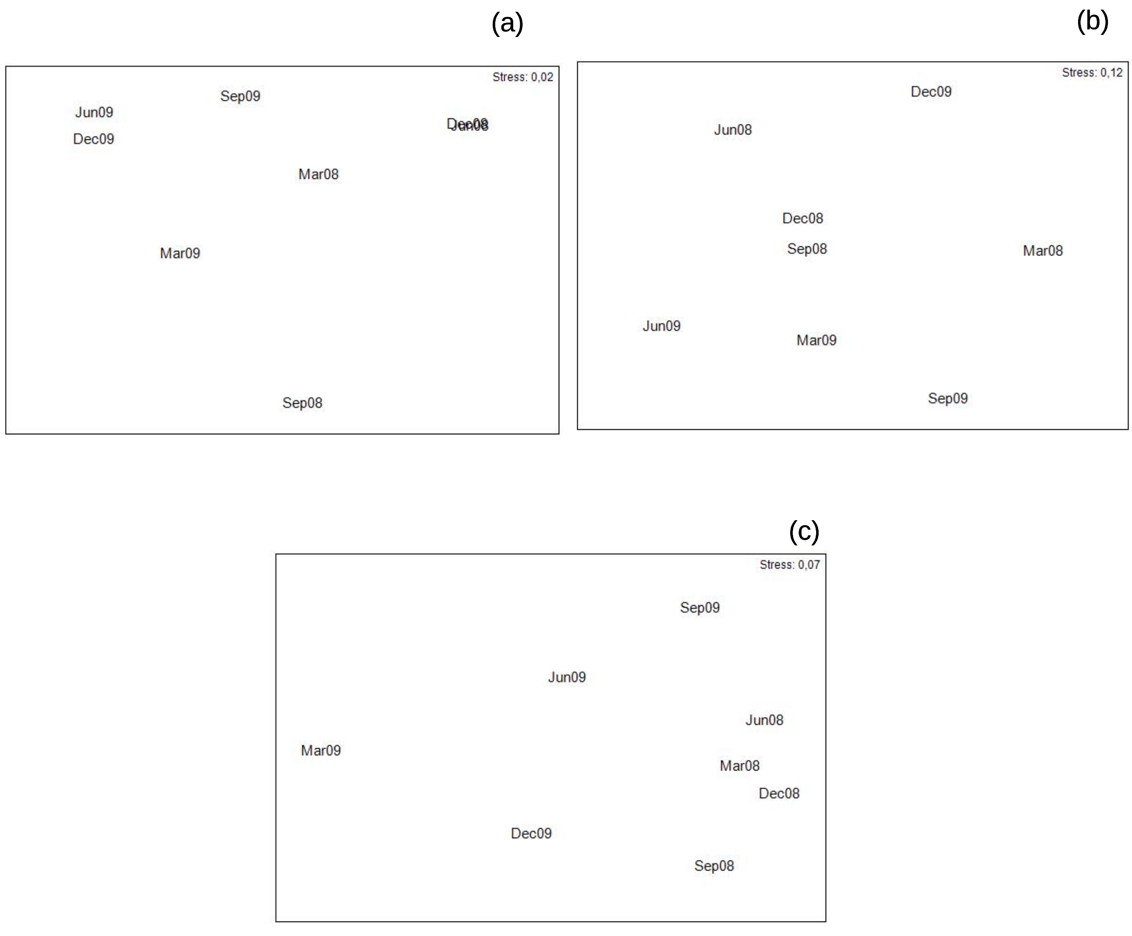

Fig. 3 Results of 2nd STAGE-MDS for the three reservoirs based on biological data collected in December (Dec), March (Mar), June (Jun) and September (Sep) of 2008 (08) and 2009 (09), (a) Serra Azul, (b) Vargem das Flores the (c) Ibirité.

Table 2. Results of ANOSIM pair wise tests between the samples of Serra Azul, Vargem das Flores, and Ibirité reservoirs; ns indicates p values >0.05.

Months	Serra Azul (R, p)	Vargem das Flores (R, p)	Ibirité (R, p)
March/08 - March/09	0.08, 0.0004	0.18, 0.001	0.21, 0.001
March/08 - June/08	0.04, ns	-0.01, ns	0.04, 0.04
March/08 - June/09	0.09, 0.002	0.04, 0.05	0.07, 0.007
March/08 - September/08	0.02, ns	0.03, ns	0.09, 0.001
March/08 - September/09	0.09, 0.001	0.06, 0.015	0.39, 0.001
March/08 - December/08	0.04, 0.025	0.06, 0.015	0.21, 0.001
March/08 - December/09	0.15, 0.001	0.10, 0.001	0.31, 0.001
March/09 - June/08	0.05, 0.018	0.20, 0.001	0.20, 0.001
March/09 - June/09	0.02, 0.094	0.22, 0.001	0.14, 0,001
March/09 - Setember/08	0.05, 0.025	0.13, 0.003	0.10, 0.001
March/09 - Setember/09	-0.01, ns	0.25, 0.001	0.13, 0.005
March/09 - December/08	0.08, 0.004	0.14, 0.001	0.07, 0.005
March/09 - December/09	0.02, ns	0.34, 0.001	0.20, 0.001
June/08 - June/09	0.06, 0.015	0.1, 0.001	0.07, 0.007
June/08 - September/08	0.05, 0.023	0.01, ns	0.05, 0.03
June/08 - September/09	0.04, 0.035	0.15, 0.001	0.22, 0.001
June/08 - December/08	0.03, 0.038	0.07, 0.015	0.21, 0.001
June/08 - December/09	0.10, 0.002	0.21, 0.001	0.26, 0.001
June/09 - September/08	0.04, 0.029	0.09, 0.004	0.11, 0.001
June/09 - September/09	0.03, 0.054	-0.02, ns	0.17, 0.001
June/09 - December/08	0.08, 0.002	0.12, 0.001	0.25, 0.001
June/09 - December/09	0.04, 0.036	0.01, ns	0.22, 0.001
September/08 - September/09	0.05, 0.014	0.13, 0.004	0.11, 0.003
September/08 - December/08	0.01, ns	0.02, ns	0.09, 0.006
September/08 - December/09	0.10, 0.001	0.18, 0.001	0.23, 0.001
September/09 - December/08	0.06, 0.008	0.16, 0.001	0.18, 0.001
September/09 - December/09	-0.01, ns	0.001, ns	0.06, 0.007
December/08 - December/09	0.10, 0.001	0.16, 0.001	0.35, 0.001

For those comparisons with a higher R value ($R>0.2$) and a significant p value ($p<0.05$) the differences were not consistent with the climatic patterns. Therefore, there was no reason for considering different Maximum Ecological Potential values for different seasons and in further analysis the mean taxa abundance was used for each sampling site.

Maximum Ecological Potential

The first axis of PCA (Fig. 4) explained 39.4% of data variability and correlated mainly with the variables total dissolved solids (0.333), TSI1 (0.326), TSI2 (0.324) and bottom substrate odour (0.329). The second PCA axis explained 17.0% of data variability and was correlated with presence of docks/boats (-0.477), roads (-0.360), pasture (-0.325) and power lines (-0.291) (Table 3). The sites selected as having less anthropogenic impact, and therefore, with the Maximum Ecological Potential, are located on the negative side of PC1 and closer to zero on PC2 (32 sites; Fig. 4).

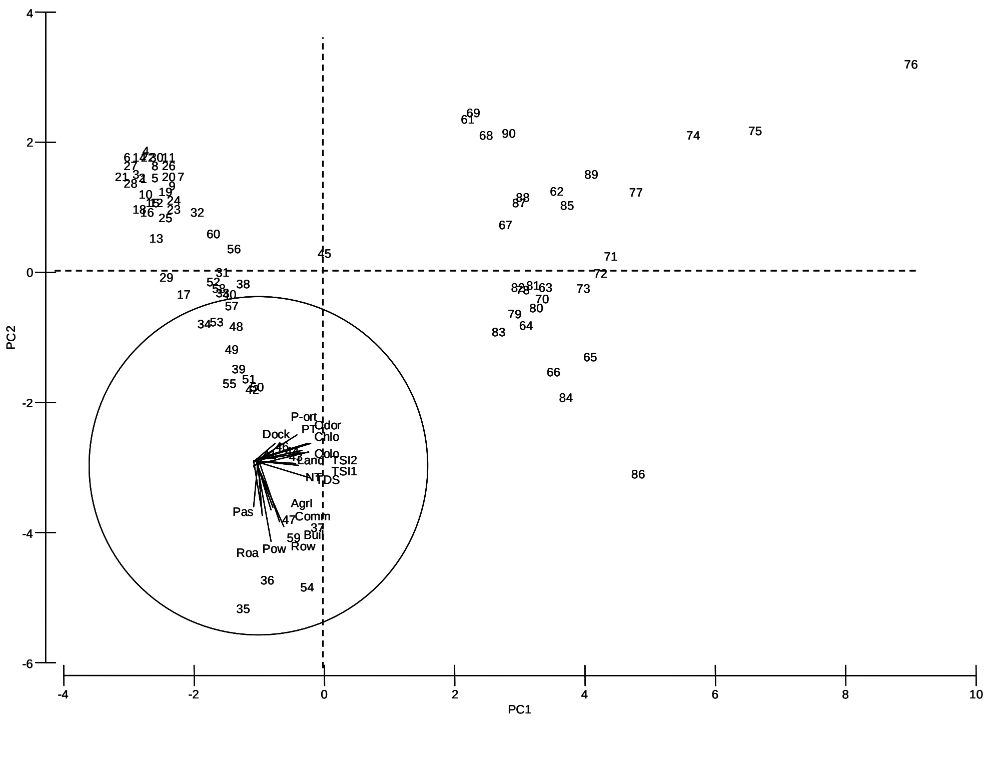

Fig. 4 Principal Component Analysis (PCA) of the 90 sites sampled in the three reservoirs based on all available pressure data (see Table 1).

Table 3. Correlation of pressure variables with the PCA axis 1 and 2.

Characterization variable	Factorial axes	
	F1	**F2**
Total Dissolved Solids (mg L^{-1})	0.333	-0.043
Chlorophyll *a (*µ*g* L^{-1})	0.267	0.079
Total nitrogen (mg L^{-1})	0.265	-0.027
Total phosphorus (µg L^{-1})	0.256	0.162
P-ortho (µg L^{-1})	0.161	0.139
Odour of bottom substrate	0.329	0.120
TSI1	0.324	0.078
TSI2	0.326	0.051
Buildings	0.160	-0.370
Commercial buildings	0.095	-0.256
Docks/boats	0.086	-0.477
Dykes	0.044	-0.270
Landfills	0.126	0.109
Roads	0.123	-0.360
Power lines	0.085	-0.291
Row crops	-0.027	-0.271
Pastures	0.026	-0.325
Agriculture	0.248	-0.016

Three of the sites selected as reference on PCA (31, 32, 56) were subsequently eliminated after observing the box plots since they showed outlier values for some pressure variables such as total phosphorus, total nitrogen, and chlorophyll *a*. After that removal all MEP sites were located in the Serra Azul reservoir. Then, we calculated the minimum and maximum acceptable values of a site with Maximum Ecological Potential for all pressure variables (Table 4).

The ordination of the biological nMDS data (stress = 0.20, 2D) confirmed that the communities of the selected MEP sites were in fact different from those of the remaining sites and therefore, were conceivably not being affected by the examined variables (Fig. 5). Taxa such as *Melanoides tuberculatus,* Oligochaeta, *Chironomus,* were found in higher proportions in the impacted sites (25.47%, 24.83% and 9.27%. respectively) and in lower proportions in sites with MEP (2.62%, 3.42%, 1.05%, respectively). The taxa

Fissimentum, Philopotamidae, Hydrobiosidae and *Procladius* were found in higher proportions in sites with Maximum Ecological Potential (5.45%, 0.04%, 0.04% and 3.16% respectively) and in small amounts or absent in impacted sites (0.24%, 0%, 0%. 0.09%, respectively). In the 28 sampling stations classified with Maximum Ecological Potential, 2.366 organisms belonging to 39 taxa (1 Mollusca, 2 Annelida and 36 Arthopoda) were collected.

Table 4. Range of acceptable values (minimum-maximum) for pressure variables in sites elected as having Maximum Ecological Potential.

Variables	**Reference**
Total dissolved solids (mg L^{-1})	16.34 - 22.10
Chlorophyll *a* (µg L^{-1})	0.13 - 3.33
Total phosphorus (µg L^{-1})	11.05 - 29.52
Total nitrogen (mg L^{-1})	0.04 - 0.10
P-ortho (µg L^{-1})	5.05 - 12.36
TSI1	29.47 - 43.91
TSI2	35.02 - 51.15
Odour of bottom substrate	1 - 2
Commercial Buildings	1 - 2
Dykes	1 - 2
Landfills	1 - 2
Roads	1 - 2
Docks/boats	1 - 2
Power lines	1 - 2
Buildings	1 - 2
Row crops	1 - 2
Pastures	1 - 2
Agriculture	1 - 2

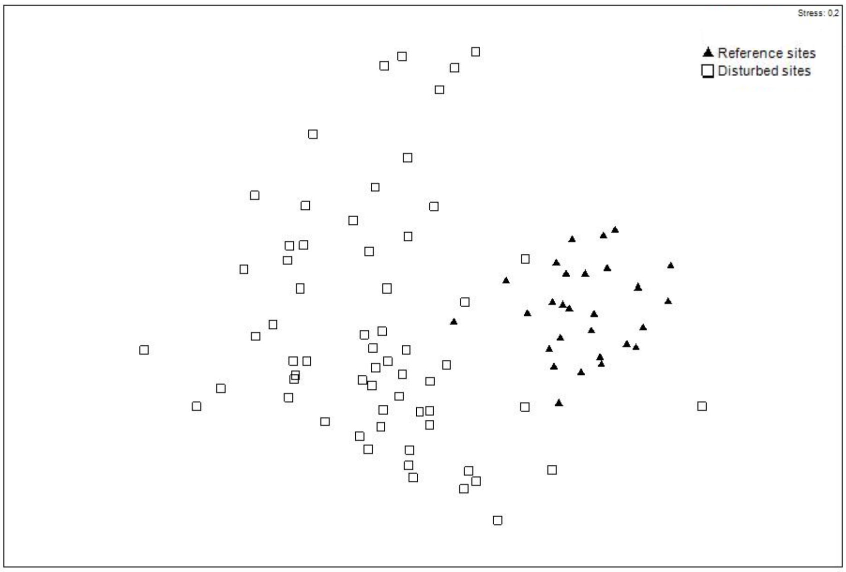

Fig. 5 Multidimensional scaling analysis based on the biological data of sites selected as having Maximum Ecological Potential and the remaining disturbed sites.

Subsets of communities in Maximum Ecological Potential

Cluster analysis results showed the existence of two local groups, G1 = 5 sites and G2 = 23 sites, within the MEP sites (Fig. 6).

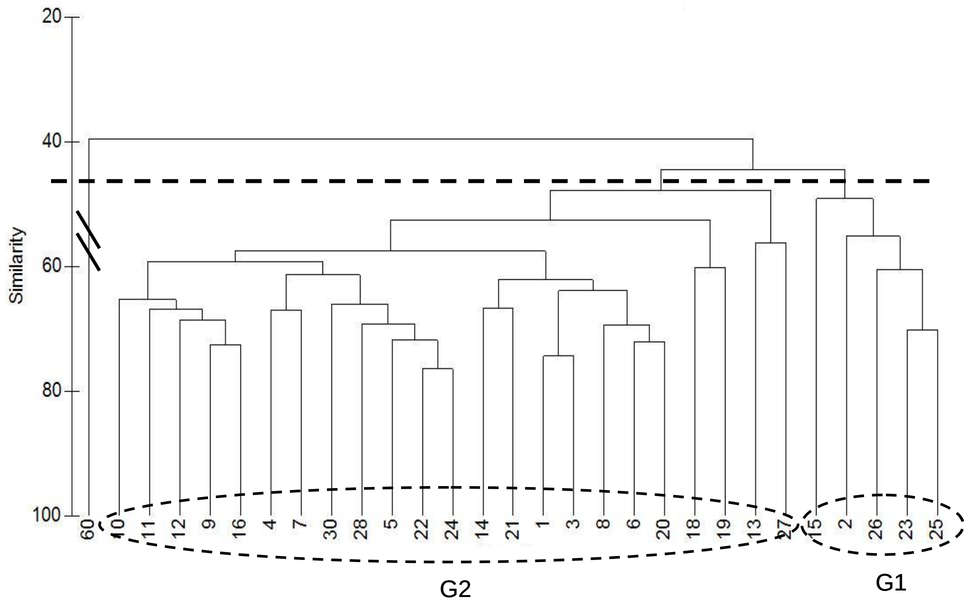

Fig. 6 Dendrogram based on the benthic invertebrates communities of sites with Maximum Ecological Potential and groups selected (G1, G2).

The ANOSIM (Global R = 0.686, p = 0.002) confirmed the significant difference between these two sub-groups (Stress = 0.21, 2D). Alternative grouping levels were tested, but the groups showed no significant differences ($p>0.05$). Site 60 was excluded owing to the fact that it was highly dissimilar from all other sites.

SIMPER analysis showed that the average Bray-Curtis similarity within each group, in terms of benthic macroinvertebrates was similar in both groups (55% for G1 and 57% for G2) (Table 5). The total abundance and the exclusive presence of the following taxa in Group 2 contributed significantly to the dissimilarity between both groups: *Ablabesmyia, Cladopelma, Aedokritus, Tanytarsus, Pseudochironomus, Alotanypus, Cryptochironomus, Stenochironomus, Parachironomus, Labrundinia, Paralauterboniela, Manoa, Chironomus*, Leptoceridae, Gomphidae and Hydrobiosidae; whereas Group 1 presented no exclusive taxa contributors to the dissimilarity between groups

Ecological indices values differed between group, with sites of Group 2 (the largest group) displaying higher values of average taxonomic richness (12.09 taxa) and Margalef's index (5.30) (Table 5) than Group 1. Both groups obtained similar Pielou evenness values (G1: 0.78; G2: 0.62).

Table 5. Average abundance of the taxa that contributed up to 99% of Bray-Curtis similarity (SIMPER analysis) between sites of the same group.

Taxa	Group 1 n=5	Group 2 n=23
Mollusca		
Gatropoda		
Thiaridae		
Melanoides tuberculatus Müller, 1774	23.14	4.82
Annelida		
Hirudinea	0.43	1.34
Insecta		
Odonata		
Gomphidae	0	0.88
Diptera		
Ceratopogonidae	0.32	2.00
Chaoboridae		
Chaoborus Lichtenstein, 1800	37.12	38.26
Chironomidae		
Tanypodinae		
Tanypus Meigen, 1803	0	7.19
Coelotanypus Kieffer, 1913	11.69	12.37
Ablabesmyia Johhansen, 1905	0	5.93
Nimbocera Reiss, 1972	0	1.25
Djalmabatista Fittkau, 1908	8.32	3.86
Procladius Skuse, 1803	7.81	5.83
Chironominae		
Tanytarsus Kieffer, 1921	0	0.20
Chironomus Meigen, 1803	0	1.99
Fissimentum Cranston & Nolte, 1996	3.18	9.81
Pelomus Reiss 1989	0	0.50
Polypedilum Kieffer, 1913	4.99	2.66
Taxonomic Richness	7.33	12.09
Total individual	3.7	12.24
Equitability Pielou's evenness	0.78	0.62
Margalef 's Richness Index	5.30	4.91
Shannon-Wiener Diversity	1.54	1.55

Abiotic typology

The results of the Stepwise Discriminant Analysis selected three descriptor variables of the bottom substrate (gravel/boulders, coarse sand, silt/clay, muck) and silt/clay/muck of the shoreline and depth, as being those that best discriminate the two biological groups of MEP sites (F = 10.66, p = 0.0001). A Jackknifed cross-validation showed that 100% of the sites in G1 and 95% of the sites in G2 are correctly assigned using the selected variables (Table 6). It can be stated that, in general, the sites belonging to Group 1 have a larger substrate (gravel, boulders and coarse substrate) and are shallower than those in Group 2 (Table 6).

Table 6. Variables selected by the stepwise forward. Discriminant Analysis and respective mean values (±DS) for the two groups Maximum Ecological Potential.

Variables	F-to-Renove	Tolerance	Group 1	Group 2
Gravel/boulders - bottom	0.525	0.786	6.26 ± 14.84	2.81 ± 11.53
Coarse sand - bottom	23.059	0.428	6.13 ± 6.23	2.27 ± 3.64
Silt, clay or muck - bottom	0.086	0.460	40.25 ± 23.88	38.93 ± 17.49
Silt, clay or muck - shoreline	43.669	0.273	1.33 ± 0.51	1.90 ± 0.30
Depth	0.525	0.786	0.61 ± 0.23	0.63 ± 0.27

Discussion

In rivers, both in temperate and tropical regions, seasonal climate variability is accompanied by changes in the communities (Sporka et al., 2006; Leunda et al., 2009; Puntí et al., 2009). These changes are known to affect the ecological assessments based on reference conditions that represent the systems only for a given season, as shown by several authors (e.g. Feio et al., 2006; Aroviita et al., 2010). However, seasonal variability observed in the two years of sampling (precipitation and temperature) was not reflected in the benthic communities of the studied reservoirs. Indeed, seasonal variability in communities was unpredictable and similar to the inter-annual variability. Other investigations, both in subtropical systems (China) and temperate systems

(Canada), showed that rainfall and the flood pulse did not influence the distribution of Chironomidae of reservoirs since they are well adapted to fluctuations in the water level (Zhang et al., 2010; Furey et al., 2006).

In this study, we considered sites with Maximum Ecological Potential those that were least impaired within our systems and data set. Additionally, the selected sites showed values that were within acceptable limits for class 1 (waters allocated to the preservation of the natural balance of aquatic communities, the preservation of aquatic environments in conservation units and entirely protected areas) according to the Brazilian legislation for water quality (CONAMA/357, CONAMA, 2005). And, as we predicted, all of the selected MEP sites belong to Serra Azul reservoir, which is located in an area of special protection with native vegetation characteristic of the cerrado forest (COPASA, 2004). This gives us additional confidence to consider them as representatives of the Maximum Ecological Potential for tropical reservoirs in the study region. Moreover, the Water Framework Directive (Directive, 2000/60/EC) recommends that for reservoirs the classification of Maximum Ecological Potential should be given when the communities are similar to those of a comparable high quality natural lake. A work performed by Ramos (2008) in the natural lakes Dom Helvécio and Águas Claras, also in Minas Gerais state, showed a species richness ranging from 12 to 23 taxa (lake and Dom Helvécio and Águas Claras lake, respectively) which is less that we found in our sites classified with Maximum Ecological Potential (39 taxa), for a comparable level of identification. Also many common taxa to our list were found in those lakes (e.g. *Coleotanypus*, *Cryptochironomus*, *Fissimentum*, *Goeldichironomus*, *Lauterboniella*, *Polypedilum*, *Procladius*, *Tanytarsus*, *Harnisch*, *Zavreliella*) and only 4 taxa were absent from our samples.

Following the construction of the reservoirs, great changes can be expected in the physical and chemical characteristics of the water and on the functional and structural composition of aquatic communities, with a reduction of the number of species and the establishment of exotic species (Horsák et al., 2009; Yanling et al., 2009). The colonization of new highly modified habitats, as in the case of reservoirs, is undertaken by highly resistant species, adapted to stagnant waters as well as generalist species, small in size, with long life cycles, and high rates in sexual maturation (Prat & Daroca, 1983; Rueda et al., 2006; Ruse, 2010). In our reservoirs, even in the selected sites with the MEP, the observed taxa richness (51 taxa, 59% Diptera) was lower than that on the river, in the same drainage basin, where 63 taxa were recorded, with Ephemeroptera, Plecoptera and Trichoptera (EPT) representing 16% of the total individuals (A. Lessa, unpublished data). In our study, the presence of the exotic species *Melanoides tuberculatus* Müller, 1774 (Thiaridae, Gastropoda) was recorded at MEP sites. Since it was first recorded in Brazil in 1967 (Rocha-Miranda & Martins-Silva, 2006) this African-Asian species has extensively invaded the tropical freshwater ecosystems, settling in various types of substrate (Dudgeon, 1989; Clementes et al., 2006). However, the densities of *M. tuberculatus* in disturbed habitats are likely to increase and may surpass the 10,000 ind m^2 (Santos & Eskinazi-Sant'Anna, 2010). In comparison with reference sites, our disturbed sites accounted for about 97% more individuals of this species. For these reasons, we still consider sites with MEP those where this species was present, but their abundance should ideally decrease.

Besides that, differences in taxa composition between MEP sites and more disturbed sites exists: Oligochaeta, the above mentioned *M. tuberculatus* and *Chironomus* represented 60% of total individuals in more disturbed sites whereas in the reference sites, they accounted only for 7%. Some genera of Chironomidae (*Manoa*, *Pseudochironomus*, *Stenochironomus*, *Zavreliella*, *Lauterboniella*, *Paralauterboniella*), Philopotamidae and Hydrobiosidae were found only in reference sites.

Several authors have showed that different Chironomidae species have different sensitivities to stress (Davies & Jackson, 2006; Arimoro et al., 2007; Roque et al., 2010). In our study, high abundance of *Polypedilum* was recorded in reference sites. However, species of this genus present a high variability of sensitivities to environmental stress (Roque et al., 2010). Therefore, and regarding taxonomic sufficiency, we think that, contrary to streams where family level is often considered sufficient for monitoring purposes (e.g. Hewlett, 2000; Feio et al., 2006; Buss & Vitorino, 2010), in reservoirs it is important to have the individuals identified to a lower taxonomic level (species).

Methods of subdividing the reference conditions are necessary in order to cover the natural variability found in the studied area and, simultaneously, must have biological relevance to make appropriate comparisons (Rawer-Jost et al., 2004). In this study, we found a good correspondence between the biological classification in two groups and some environmental descriptors. Variables related to bottom substrate and substrate of the shoreline and depth correctly discriminate 96% of the 28 reference sites in their respective biological groups. This allowed the construction of an abiotic typology relevant to the invertebrates belonging to the littoral zone of reservoirs. This typology will enable the future establishment of appropriate comparisons between the communities of new sites to be assessed and the values of reference condition set by sites with Maximum Ecological Potential.

Camargo et al. (2005) observed that, similarly to streams and rivers (see Bailey et al., 1998; Rawer-Jost et al., 2004), sediment is a key factor that determines the spatial distribution of invertebrates in reservoirs. This fact is corroborated in our study, where substrate was responsible for the differential distribution of Chironomidae genera between the two biological groups at in the reference sites.

We found depth to be an important discriminator of reservoir benthic invertebrate in our study sites. This is in accordance with other authors that have identified depth, as an important factor in structuring the communities of reservoirs (Verneaux et al., 2004; Rossaro et al., 2007; Panis et al., 1996). Also, studies undertaken in both Spanish (Prat et al., 1991) and Brazilian reservoirs (Moretto et al., 2003; Moreno & Callisto, 2006) found that the presence of Chironomidae in deep areas is, in general, very poor.

In conclusion, we found within our reservoirs, sites that can be considered in Maximum Ecological Potential that were useful to establish a baseline for a future bioassessment tool to monitor the ecological quality of tropical reservoirs of the study area. This does not prevent the search of better references even though we think that our MEP sites are in fact in a privilege condition regarding land use and water quality, when compared to other reservoirs in the Brasil. An investment in the area of taxonomy, leading to the identification to species level could be the advent of a more sensitive and accurate assessment system.

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APPENDIX I

Taxon	Maximum Ecological Potential Abundance n	Maximum Ecological Potential Frequency (%)	Impacted sites Abundance n	Impacted sites Frequency (%)
Mollusca				
Gatropoda				
Prosobranchia				
Mesogastropoda				
Thiaridae				
Melanoides tuberculatus Müller, 1774	62	2.62	3078	25.47
Planorbiidae				
Biomphalaria straminea Dunker, 1848	0	0	18	0.14
Ampullariidae				
Pomacea haustrum Reeve, 1856	0	0	10	0.08
Bivalvia				
Corbiculidae				
Corbicula fluminea Müller, 1774	0	0	34	0.28
Annelida				
Hirudinea	17	0.71	167	1.38
Oligochaeta	81	3.42	3000	24.83
Insecta				
Ephemeroptera				
Polymirtacyidae	9	0.38	12	0.09
Baetidae	2	0.08	2	0.01
Leptoceridae	5	0.21	15	0.12
Odonata				
Gomphidae	12	0.50	6	0.04
Trichoptera				
Odontoceridae	0	0	1	0.04
Hydrophilidae	1	0.04	1	0.04
Philopotamidae	1	0.04	0	0
Hydrobiosidae	1	0.04	0	0
Coleoptera				
Elmidae	1	0.04	14	0.11
Acari				
Hidracarina	1	0.04	0	0
Diptera				
Chaoboridae				
Chaoborus Lichtenstein, 1800	1345	56.84	2914	24.12
Simuliidae	0	0	8	0.06
Ceratopogonidae	31	1.31	36	0.29
Tanypodinae				
Nilothauma Kieffer, 1921	1	0.04	1	0.04
Labrundinia Roback, 1987	1	0.04	0	0

Cont. Appendix I

Coelotanypodini				
Coelotanypus Kieffer, 1913	165	6.97	367	3.03
Pentaneurini				
Ablabesmyia Johhansen, 1905	48	2.02	21	0.17
Procladiini				
Djalmabatista Fittkau, 1968	73	3.08	27	0.22
Procladius Skuse, 1803	75	3.16	12	0.09
Tanypodini				
Tanypus Meigen, 1803	145	6.12	443	3.66
Chironominae				
Chironomini				
Dicrotendipes Kieffer, 1913	0	0	2	0.01
Beardius Reiss & Sublette, 1985	0	0	1	0.04
Aedokritus Roback, 1958	6	0.25	480	3.97
Chironomus Meigen, 1803	25	1.05	1120	9.27
Cladopelma Kieffer, 1921	6	0.25	2	0.01
Cryptochironomus Kieffer, 1918	2	0.08	1	0.04
Fissimentum Cranston & Nolte, 1996	129	5.45	29	0.24
Goeldchironomus Fittkau, 1965	1	0.04	36	0.29
Harnischia Kieffer, 1921	0	0.00	0	0
Lauterboniella Lenz, 1941	7	0.29	0	0
Paralauterboniella Lenz, 1941	1	0.04	0	0
Pelomus Reiss, 1989	10	0.42	78	0.14
Polypedilum Kieffer, 1913	66	2.78	5	0.04
Stenochironomus Kieffer, 1919	1	0.04	0	0
Zavreliella Kieffer, 1920	14	0.59	0	0
Alotanypus Roback, 1971	2	0.08	0	0
Parachironomus Lenz, 1921	1	0.04	2	0.08
Pseudochironomini				
Manoa Fittkau, 1963	1	0.04	0	0
Pseudochironomus Mallock, 1915	2	0.08	0	0
Tanytarsini				
Tanytarsus van der Wulp,1984	8	0.34	97	0.80
Caladomyia Sawedal, 1981	7	0.29	19	0.19

CAPÍTULO 5

Development and test of a statistical model for the ecological assessment of tropical reservoirs based on benthic macroinvertebrates

Joseline Molozzi[1], Maria João Feio[2], Fuensanta Salas[3],

João Carlos Marques[2] & Marcos Callisto[1]

Reservatório de Ibirité

1- Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, ICB, Universidade Federal de Minas Gerais, Brasil
2- Intituto do MAR-CMA, Departamento de Ciências da Vida, Universidade de Coimbra, Portugal
3- TRAGSA, Murcia, Espanha

Submeter à Aquatic Ecology

Development and test of a statistical model for the ecological assessment of tropical reservoirs based on benthic macroinvertebrates

Joseline Molozzi, Maria João Feio, Fuensanta Salas,

João Carlos Marques & Marcos Callisto

Abstract: Reservoirs are heavily modified lentic areas. However, it is important to keep their chemical status and aquatic communities with a good ecological potential status. In the present work we test the value of an assessment tool based on the structure of benthic macroinvertebrate communities to evaluate the Ecological Potential (EP) of tropical reservoirs. For that, we designed a conceptual assessment scheme based on the Reference Condition Approach and developed a statistical model based on 28 sites classified as having maximum ecological potential from three reservoirs in the region of Belo Horizonte, Minas Gerais, Brazil. Sixty-two disturbed sites were used to test the model. A classification system based on 3 EP classes was found the best option and traduced different levels of total Nitrogen, orthophosphates, turbidity, TDS and trophy. This study confirmed the utility of benthic macroinvertebrate as a biological quality element in reservoirs and the statistical model applied was effective in providing a measure of the ecosystem health of the reservoirs. As a further improvement, the level of taxonomic resolution for some groups, such as the chironomids, could be increased, as species level may provide a better discrimination of intermediate degradation levels.

Keywords: Benthic macroinvertebrates, ecological potential assessment, reference condition approach, tropical reservoirs.

Introduction

Nowadays most biomonitoring methods of aquatic systems are based on the Reference Condition Approach, where the expected biological community of a site is extrapolated from the environmental conditions and can be compared with the community found at reference sites (Reynoldson et al., 1997; Directive 2000/60/EC; Bailey et al., 2004; Sttodart et al., 2006). Following this philosophy, the first predictive models were developed for rivers based on macroinvertebrate communities (Wright et al., 1996; Reynoldson et al., 1995,1997; Wright, 2000). These predictive methods, such as the RIVPACS (River Invertebrate Prediction Classification System; Wright, 2000; Clarke, 2000; Reynoldson & Wright, 2000), BEAST (Benthic Assessment of Sediment; Reynoldson et al., 1995; 2000) and ANNA (Assessment by Nearest Neighbor Analysis; Linke et al., 2005) allow for the direct classification of water quality and are used to monitor the quality of a site over time. More recently, other biological elements have been used in the development of models (e.g. Joy & Death, 2002; Kennard et al., 2006; Feio et al., 2007a) and the methodology has also been adapted to other systems, such as lakes and swamps (e.g. Johnson & Sandin, 2001; Tall et al., 2008).

Traditionally the assessment of reservoirs in lakes have been done mostly through the analysis of physical and chemical parameters and measurements of chlorophyll *a* (e.g. Canfield & Bachmann, 2001; Carrillo et al., 2003) since the main concern was to avoid algae blooms and maintain a reasonable water quality for domestic and agricultural purposes. In Brazil, some indices based on chemical parameters are presently used, such as the Water Quality Index (WQI) (CETESB, 2006) and the Trophic State Index (TSI) (Carlson, 1977 modified by Toledo et al., 1983) but biological elements are not considered. Recently in Europe, and especially after the European Water Framework

Directive (WFD, 2003; Directive 2000/60/EC), phytoplankton community (identification and quantification) based approaches started to be developed (e.g. Elliott et al., 2005; Cabecinha et al., 2009) and there are also other studies focusing on the potential of using of fishes as bioindicators for reservoirs (Adams et al., 1999; Terra & Araújo, 2011).

Freshwater macroinvertebrates are widely used as bioindicators in running waters since these organisms have limited mobility, are more sensitive to local disturbances than pelagic organisms, are capable of detecting structural changes and habitat loss, and species have different degrees of stress tolerance (Hellawell, 1977; De Pauw & Vanhooren, 1983; Karr, 1991; Barbour et al., 1996). However, only recently, some efforts have been made, in temperate zones, to develop biological evaluation tools for lakes and reservoirs based on benthic invertebrates, (Johnson & Sandin, 2001; Blocksom et al., 2002; Martin & Rippey, 2008; O'Toole et al., 2008; Peeters et al., 2009).

Thus, the aims of the present study are: 1) to avaliable that benthic macroinvertebrate communities in littoral areas (in shallow waters) are an useful biological indicator of the degradation of tropical reservoirs; and 2) to propose and test a conceptual scheme and a statistical model for this evaluation based on the Reference Condition Approach.

The approach proposed here, eventhough using different statistical analyses, follows closely Canadian BEAST models (Reynoldson et al., 1995, 1997), since it considers that a site is disturbed when its community is different from the communities found in reference sites, or in this case, with Maximum Ecological Potential, even though the statistical methods are different from the one described for the BEAST models. Our model was based on 28 sites previously classified as having maximum ecological potential (Molozzi et al., submitted) from three reservoirs in the region of Belo

Horizonte, Minas Gerais State, Brazil. Sixty-two disturbed sites from those reservoirs were used to test if the model explains the existing gradient of human degradation.

Materials and Methods

Study area

This work was based on data collected in three reservoirs (Ibirité, Vargem das Flores and Serra Azul) , located in Paraopeba river watershed, an affluent of the São Francisco river in Minas Gerais State, south-eastern Brazil. In this region the climate is considered tropical sub-humid (Cwb), with summer rains (November to April) and a dry winter (May to October). Average annual temperature is c. 20^{o}C (Moreno & Callisto, 2006) (Fig. 1).

The Ibirité reservoir (20°01’13.39” S ; 44°06’44.88”W) was built in 1968 at an altitude of 773 m a.s.l. This reservoir has an area of 2.8 km^2, a volume of 15,423,000 m^3 and an average depth of 16 m. The hydrographic basin of the Ibirité Reservoir extends over two municipalities, Ibirité (148,535 inhabitants) and Sarzedo (23,282 inhabitants).

The Vargem das Flores reservoir (19º54’25.06’’ S; 44º09’17.78” W) has a surface area of 4.9 km^2, contains 37,000,000 m^3 of water and has a maximum depth of 18 m. The maximum height of sill spillway is 837 m and the reservoir as a hydraulic retention time of 365 days. About 12,3 ha of the area around the reservoir were transformed into a special protected area of the state of Minas Gerais in 2006 (COPASA, 1980a).

The Serra Azul reservoir (19°59’24.92’’ S; 44°20’46.74” W), located at an altitude of 760 m a.s.l., has a water surface of 7.5 km^2, water volume of 88,000,000 m^3 and a maximum depth of 40 m. It has been operating for approx. 30 years as a source of drinking water to the metropolitan region of the State’s capital (ca. 4.8 million people).

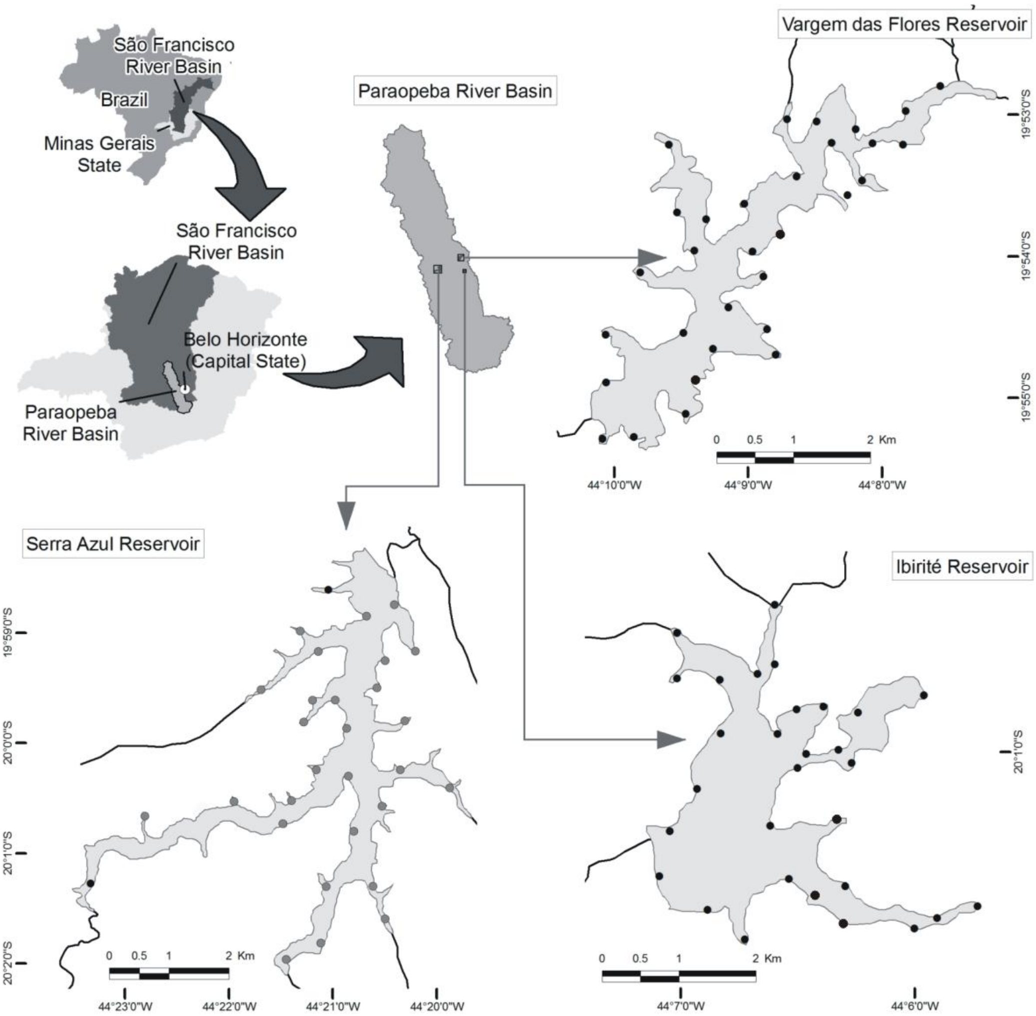

Fig. 1 Location of the reservoirs Vargem das Flores, Ibirité and Serra Azul in the catchments of the Paraopeba River, Minas Gerais, Brazil and distribution of the sampling sites with Maximum Ecological Potential (grey dots) and disturbed sites (black dots) in the reservoirs.

The reservoir has a hydraulic retention time of 351 days. Surrounding this reservoir there is also a special protected area established in 1980 with an area of 27.000 ha (COPASA, 2004). Inside this area, 3.2 ha belong to the COPASA, the industry that explores the reservoir (COPASA, 1980b), and no tourism or fisheries are allowed.

Macroinvertebrate sampling

The macroinvertebrate samples were collected in the littoral region of the reservoirs. Samples were collected quarterly over a period of two years 2008 and 2009 (March, June, September, December) with an Eckman-Birge dredge (0.0225 m^2), at a median depth of 3.92 m. The material collected was fixed in formaldehyde 70% and subsequently identified to family and genus level (Chironomidae) (Peterson, 1960; Trivinho-Strixino, 2011; Merritt & Cummins, 1996; Mugnai et al., 2010).

Abiotic data

At each sampling occasion the following water physical and chemical parameters were measured using an YSI Model Multiprobe: water temperature, dissolved oxygen, conductivity, turbidity and pH. Additionally, sub-surface water samples were collected with a Van Dorn type cylinder for subsequent measurement of total nitrogen (TN), total phosphorus (TP) and orthophosphates (PO_4), in accordance with "Standard Methods for the Examination of Water and Wastewater" (APHA, 1992). The concentration of chlorophyll *a* (Chla) was obtained according to Golterman et al. (1978) (Table 1). The depth of the euphotic zone (S) was determined based on the readings of the Secchi disc.

The Carlson (1977) trophic state index (TSI1), modified by Toledo et al., (1983), and the Trophic State Index proposed by CETESB (2000) (TSI2) were calculated for all sites. Each index is composed by sub-indices, which are then weighted to obtain a final value of the trophic status. The TSI1 is calculated through the formula:

a) $TSI1 = TSI(S) + 2 *[TSI(TP) + TSI(P0_4) + TSI(Chla)/7]$,

and the sub-indices are obtained as follows:

$$TSI(S) = TSI(S) = 10 * (6 - (0,64 + \ln S) / \ln 2))$$

$$TSI(TP) = 10 * (6 – (\ln (80,32 / TP) / \ln 2))$$

$$TSI(PO4) = 10 * (6 – (\ln (21,67 / PO4) / \ln 2))$$

$$TSI(Chla) = 10 * (6 – (2,04 – 0,695 \ln Chla) / \ln 2))$$

The TSI2 is calculated through the formula:

b) $TSI2= [TSI(TP) + TSI(Chla)]/2$,

and the sub-indices are obtained through the expressions:

$$TSI(TP) = 10 * (6 – ((1.77-0.42)* \ln (TP) / \ln 2))$$

$$TSI(Chla) = 10 * (6 – (0.92 – 0.34)* \ln Chla) / \ln 2))$$

TSI1 values ranging from 0 to 44 correspond to oligotrophic, 44–54 to mesotrophic, and > 54 to eutrophic waters. TSI2 values ranging from 0 to 23 correspond to ultraoligotrophic, 24–44 to oligotrophic, 44–54 to mesotrophic, 54–74 to eutrophic, and > 74 hypereutrophic conditions.

The relative abundance of gravel/boulders, coarse sand and silt/clay/muck substrate types in the bottom of the reservoir and silt/clay/muck in the shoreline were also assessed at each site, as they are needed to determine the typology of test sites (variables described in Table 2).

Table 1. Abiotic pressure variables used to evaluate the models sensitivity to anthropogenic disturbance.

Abiotic variables	
Total Dissolved Solids (TDS; mg L^{-1})	Field measurement (YSI)
Chlorophyll *a* (µg L^{-1})	Analysis according to Golterman et al. (1978)
Total nitrogen (TN; mg L^{-1})	Analysis according to APHA (1992)
Total phosphorus (TP; µg L^{-1})	Analysis according to APHA (1992)
P-orthophosphates (P-ortho; µg L^{-1})	Analysis according to APHA (1992)
Odour of bottom substrate	Field observation, categories: 1(none), 2(H_2S), 3(anoxic), 4(oil), 5 (chemical), 6(other) - USEPA (2007)
TSI1	Trophic index1; Based on Carlson (1977), modified by Toledo et al. (1983)
TSI2	Trophic index2; Based on CETESB (2000)
Buildings (%)	Field observation: 1=absent (0%), 2=sparse (<10%), 3=moderate (10 - 40%), 4=heavy (40 - 75%), 5=very heavy (>75%), USEPA (2007)
Commercial buildings (%)	Idem
Docks/boats (%)	Idem
Dykes (%)	Idem
Landfills (%)	Idem
Roads (%)	Idem
Power lines (%)	Idem
Row crops (%)	Idem
Pasture (%)	Idem
Agriculture (%)	Idem

Table 2. Discriminant variables description and respective mean values (±SD) in the two reference groups.

Variable	Description and Source	Group 1	Group 2
Gravel/boulders (2-4000 mm)-bottom	Field observation. Categories: 1=Absent (0%), 2=(<0-20%), 3=(20-60%), 4=(<60%)	6.26 ± 14.84	2.81 ± 11.53
Coarse sand (0.50-2 mm) - bottom	Field observation. Categories: 1= 0-15%), 2= 15-35%, 3= 35-45%, 4= <45%	6.13 ± 6.23	2.27 ± 3.64
Silt, clay or muck (<0.062mm) - bottom	Field observation. 1= 0-15%, 2= <15-35%), 3=(<35-45%), 4=(<45%)	40.25 ± 23.88	38.93 ± 17.49
Silt, clay or muck (0.062-2 mm) - shoreline	Field observation. Categories: 1=0-15%), 2=(<15-35%), 3=(<35-45%), 4=(<45%)	1.33 ± 0.51	1.90 ± 0.30
Depth	Field measurement (Sonar)	0.61 ± 0.23	0.63 ± 0.27

Maximum ecological potential sites and test sites

Twenty-eight sites were previously classified with Maximum Ecological Potential (MEP; Molozzi et al., *submitted*) based on the evaluation of the abiotic pressures (i.e. land use, water chemistry and physics, hydromorphology) affecting the littoral zone, transition zone and riparian zone and later on the integrity of the biological communities. Most of the sites with MEP come from the Reservoir of Serra Azul, which is located in an area of permanent protection COPASA-MG.

Two types, or distinct sets of sites with Maximum Ecological Potential were defined based on the biological communities collected in 8 occasions, over two years, at each site, and abiotic characteristics: in G1 the community is dominated by the Diptera *Chaoborus*, *Djalmabatista*, *Procladius* and *Polypedilum* and the sites have larger substrate (gravel, boulders and coarse substrate) and shallow depth (mean depth 0.61 ± 0.23); in G2 the community is dominated by the Diptera Ceratopogonidae, *Tanypus*, *Coelotanypus*, *Ablabesmyia* and *Fissimentum* and the sites are characterized by small substrate size (when compared to G1), and the shoreline is composed of a greater percentage of silt, clay or muck , and are deeper that those of G1 (mean depth 0.63 ± 0.27) (Molozzi et al., *submitted*). Table 2 shows the mean values for typological variables, those that best discriminate the two groups of sites with MEP (96% correct discrimination after cross validation) according to previous work (Molozzi et al., *submitted*).

On the other hand, 62 disturbed sites mainly located in the Serra Azul, Vargem das Flores and Ibirité reservoirs were used to test the model.

Data analysis

Model construction and assessment of test sites

The conceptual approach followed is shown schematically in the flow diagram (Fig. 2). The first step consists on building a classification system based on the within groups Bray-Curtis dissimilarity of MEP sites communities using the SIMPER routine (Clarke & Warwick, 2001; Primer 6). Biological data was previously averaged by site and transformed by square root. The remaining gradient (100-x) was divided by two or three, in order to obtain a 3- or 4-classes quality system, to test for the best classification system. The 4-classes quality system, also used by Reynoldson et al., (1997) and Feio et al., (2007a,b), is composed by: class 1 (Equivalent to Maximum Ecological Potential), class 2 (Moderately Different from Maximum Ecological Potential), class 3 (Different from the Maximum Ecological Potential) and class 4 (Very Different from the Maximum Ecological Potential). Alternatively, we created a 3-classes system where: class 1 is Equivalent to Maximum Ecological Potential, class 2 means that the site is Different from the Maximum Ecological Potential and class 3 means that the site is Very Different. Each class corresponds therefore to an interval of similarity to the MEP group. Although for streams and rivers, 5-or 4-class systems are more common, we think that for naturally poor systems (for invertebrate communities) such as the reservoirs, these may be too many classes to show the reflex of disturbance in the communities.

The second step consists on determining the abiotic similarity from one site to the different sub-groups of MEP in order to make the most appropriate comparison. So, in practice we have two models, one for each type. This step is accomplished with a complete discriminant analysis (Systat 13.0, Hair et al., 1998) using as discriminating variables, the typological variables, i.e., the abiotic variables that characterize and distinguish the reference groups (Table 2), according to previous work (Mollozi et al., *submitted*). In this case we have only two reference groups defined for these tropical reservoirs.

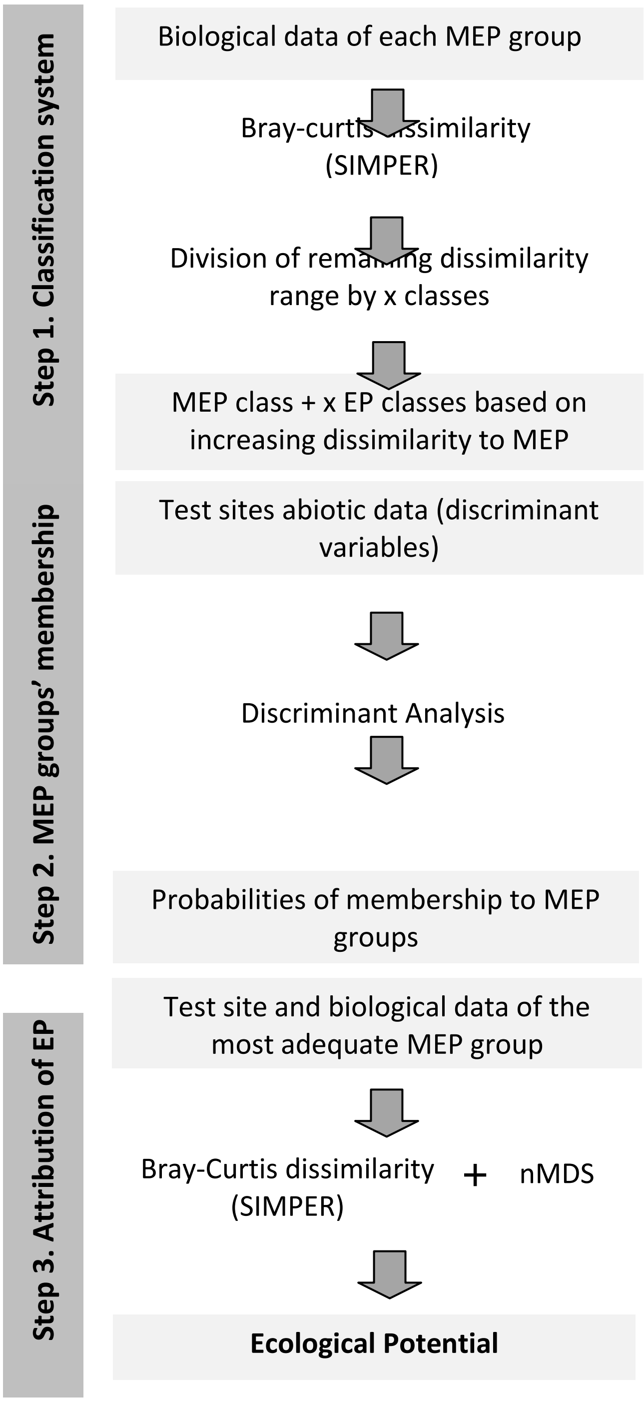

Fig. 2 Methodology used to develop the model for the ecological assessment of the reservoirs, using benthic invertebrates of region littoral. MEP, stands for Maximum Ecological Potential and EP for Ecological Potential.

In the third step, the biotic data from each test site is compared to the adequate MEP sub-group (determined in step 1) by calculating their average Bray-Curtis dissimilarity through a SIMPER analysis (Clarke & Warwick 2001; PRIMER 6 Version

6.0, Ltd., 2004). The dissimilarity between each test site and MEP sub-group was also visually inspected with a non-metric Multi-Dimensional Scaling analysis (nMDS, PRIMER-6 Version 6.0, Ltd., 2004). Finally, a quality class is attributed to test sites according to their dissimilarity and according to the classification system constructed in step 1. For each test site we obtained therefore two quality classifications according to the system being tested (3- or 4- classes quality system).

Evaluation of model response to anthropogenic disturbance

In order to determine which is the most useful quality system (3- or 4- classes), we repeated all the following tests for both systems. First, to check that the level of abiotic degradation is different between classes we used a PERMANOVA test with 999 permutations (Permutational Multivariate Analysis of Variance; Anderson, 2001a,b; Anderson & Braak, 2003; Anderson et al., 2008; software package PERMANOVA + for PRIMER, 2006, with normalized pressure data). This routine is a multivariate permutational non-parametric test, analogue to the univariate ANOVA. We also used the PERMANOVA to check that each quality class corresponded to similar levels of abiotic degradation classes when using the two different reference groups, i.e., if class 1 atributed by the model based on reference group A was similar in terms of abiotic degradation, to class 1 attributed by the model based on reference group B, and so on.

Using the Box-and-Whisker plots, we evaluated graphically if there was a progressive increase of anthropogenic degradation of test sites with the increase of class, for each pressure variable measured, i.e., if sites with class 2 had in fact a higher level of degradation than sites with class 1, and so on (Statistic 7.0).

Then, in order to chek if the distribution of sites by quality classes corresponds to differences in overall disturbance we performed a Canonical Analysis of Principal Coordinates (CAP) on normalized pressure data (Clarke & Warwick, 2001) (PERMANOVA + for PRIMER 2006). The CAP analysis provides a constrained ordination that maximizes the differences among a *priori* groups (Anderson & Braak, 2003), which are in our case the quality classes. It also shows the strength of the association between the multivariate data cloud (based on sites pressures) and the hypothesis of differences between quality classes. Additionally it calculates the probability associated with differences between multivariate groups, in the form a miss-classification error using the "leave one out allocation of the observation groups" approach. We therefore used it to compare the percentage of correct classifications to the class attributed by the model, in the 3- and 4-classes systems. Finally, to find which pressures characterize the most the differences between classes we superimposed vectors corresponding to Spearman correlations of individual pressures with the CAP axes.

Results

In the 62 test sites, 12,059 organisms from 35 taxa (4 Mollusca, 2 Annelida and 29 Arthopoda) (Table 3) were sampled over two years and eight sampling occasions. After the complete discriminant analysis, based on the abiotic predictors (typological variables), 34 of those sites showed a higher probability of belonging to G1 and 28 to the G2.

Table 3. List of taxa collected in Maximum Ecological Potential sites and test sites along the two years of sampling and respective total abundance (number of larvae collected) and frequency (% of taxa present to total abundance at each sampling site).

Taxon	Maximum Ecological Potential (n =28)	Impacted sites (n=62)

	Abundance (n)	Frequency (%)	Abundance (n)	Frequency (%)
GASTROPODA				
Thiaridae - *Melanoides tuberculatus* Müller, 1774	62	2.62	3078	25.47
Planorbiidae - *Biomphalaria straminea* Dunker, 1848	0	0	18	0.14
Ampullariidae - *Pomacea haustrum* Reeve, 1856	0	0	10	0.08
BIVALVIA				
Corbiculidae - *Corbicula fluminea* Müller, 1774	0	0	34	0.28
ANNELIDA				
Hirudinea	17	0.71	167	1.38
Oligochaeta	81	3.42	3000	24.83
EPHEMEROPTERA				
Polymirtacyidae	9	0.38	12	0.09
Baetidae	2	0.08	2	0.01
Leptoceridae	5	0.21	15	0.12
ODONATA				
Gomphidae	12	0.50	6	0.04
TRICHOPTERA				
Odontoceridae	0	0	1	0.04
Hydrophilidae	1	0.04	1	0.04
Philopotamidae	1	0.04	0	0
Hydrobiosidae	1	0.04	0	0
COLEOPTERA				
Elmidae	1	0.04	14	0.11
ACARI				
Hydracarina	1	0.04	0	0
DIPTERA				
Chaoboridae - *Chaoborus* Lichetenstein, 1800	1345	56.84	2914	24.12
Simuliidae	0	0	8	0.06
Ceratopogonidae	31	1.31	36	0.29
Chironomidae				
Tanypodinae				
Labrundinia Roback, 1987	1	0.04	0	0

Cont. Table 3

Nilothauma Kieffer, 1921	1	0.04	1	0.04

Coelotanypus Kieffer, 1913	165	6.97	367	3.03
Ablabesmyia Johhansen, 1905	48	2.02	21	0.17
Djalmabatista Fittkau, 1968	73	3.08	27	0.22
Procladius Skuse, *1803*	75	3.16	12	0.09
Tanypus Meigen, 1803	145	6.12	443	3.66
Chironominae				
Dicrotendipes Kieffer, 1913	0	0	2	0.01
Beardius Reiss & Sublette, 1985	0	0	1	0.008
Aedokritus Roback, 1958	6	0.25	480	3.97
Chironomus Meigen, 1803	25	1.05	1120	9.27
Cladopelma Kieffer, 1921	6	0.25	2	0.01
Cryptochironomus Kieffer, 1918	2	0.08	1	0.04
Fissimentum Cranston & Nolte, 1996	129	5.45	29	0.24
Goeldchironomus Fittkau, 1965	1	0.04	36	0.29
Harnischia Kieffer, 1921	0	0,00	0	0
Lauterboniella Lenz, *1941*	7	0.29	0	0
Paralauterboniella Lenz, 1941	1	0.04	0	0
Pelomus Reiss, 1989	10	0.42	78	0.14
Polypedilum Kieffer, 1913	66	2.78	5	0.04
Stenochironomus Kieffer, 1919	1	0.04	0	0
Zavreliella Kieffer, 1920	14	0.59	0	0
Alotanypus Roback, 1971	2	0.08	0	0
Parachironomus Lenz,1921	1	0.04	2	0.08
Manoa Fttikau, 1963	1	0.04	0	0
Pseudochironomus Mallock, 1915	2	0.08	0	0
Tanytarsus van der Wulp,1984	8	0.34	97	0.80
Caladomyia Sawedal, 1981	7	0.29	19	0.19

Five sites had a similar probability of belonging to the two groups (e.g. Site 76 - 48% chance of belonging to group 1 and 51% chance of belonging to group 2), and therefore, they were ordinated according to the MEP sites from both groups through an nMDS, and finally compared to the closest group (Fig. 3).

The SIMPER analysis showed that the average Bray-Curtis similarity of benthic macroinvertebrates communities was similar in both MEP sub-groups (55.28% - 57% in G1 - G2). Therefore, the division of the remaining dissimilarity gradient, to form 3 or 4 quality classes, resulted in similar intervals (Table 4).

The calculation of Bray-Curtis dissimilarity between each test site and the respective MEP sub-group and posterior allocation to the previously established classes resulted, for the 3-classes quality system, as following: 5 sites (14.7%) were classified as Different from MEP quality status (Class 2), and 29 sites (85.30%) classified as Very Different from MEP (Class 3) for G1. For G2, 20 sites (71.42%) were classified as Different to the MEP (Class 2), and 8 sites (28.58%) classified as Very Different from MEP (Table 5).

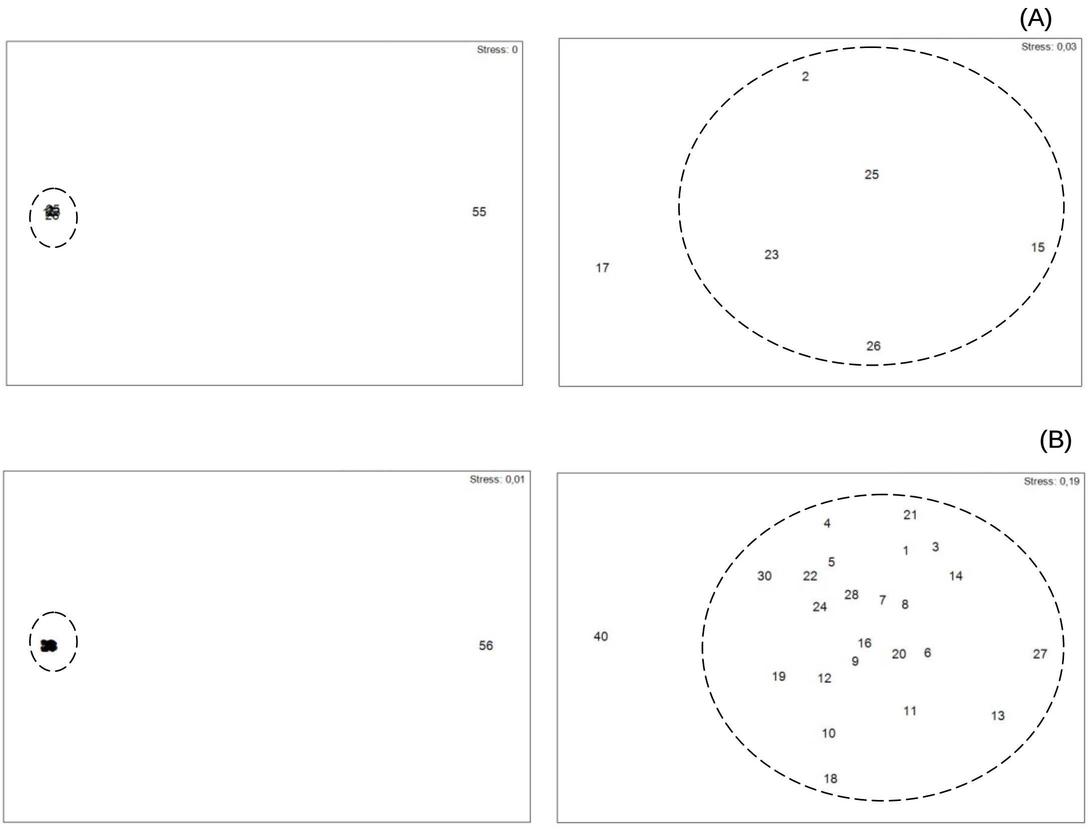

Fig. 3 nMDS analysis showing segregation between test sites and the Maximum Ecological Potential groups, located inside the dashed circles; (A) Group G1 and (B) Group G2.

For the 4-classes quality system of water quality assessment, 2 sites (5.88%) were classified as Moderately Different from MEP (Class 2), 16 sites (47.06%) were classified as Different from MEP (Class 3), and 16 sites (47.06%) classified as Very

Differente from MEP (Class 4) for G1. For G2, 9 sites (32.14%) were classified as Moderately Different from MEP (Class 2), 14 sites (50%) were classified as Different from MEP (Class 3), and 5 sites (17.86%) were attributed Class 4 (Table 5).

The PERMANOVA's demonstrated that: 1) for both 3-and 4- classes quality systems there are no differences between G1 and G2 for the same class (Pseudo-$F_{2.719}$ = 3.89, p = 0.571, Pseudo-$F_{2.719}$ = 2.89, p = 0.363, proportion p = 0.05, for 4- and 3- classes quality system); and 2) for both 3- and 4- classes systems, there were significant differences ($p<0.05$ or $p<0.001$) between classes concerning the abiotic degradation of sites (results in Table 6). However, there were no significant differences between the degradation classes 2 (moderately different from MEP quality) and 3 (different from MEP) and b 4 (different to very different from MEP), for the 4- classes system.

Table 4. Quality class intervals, for a 3- and 4- classes classification systems, based on Bray-Curtis dissimilarity.

3-classes quality system			4-classes quality system		
	G1	**G2**		**G1**	**G2**
1- Equivalent to MEP	≥ 44.72	≥ 43.00	1- Equivalent to MEP	≥44.72	≥ 43.00
2- Different from MEP	< 44.73 - 66.99	< 43.01 - 64.49	2- Moderately Different from MEP	<44.73 - 59.65	<43.01 - 57.30
3- Very Different from MEP	≤ 67.00	≤ 64.50	3- Different from MEP	<59.66 - 74.51	<57.31 - 71.62
			4- Very Different from MEP	≤ 74.52	≤71.63

The Box-Whisker plots showed increasing values of abiotic degradation for both 3- and 4-classes classification systems and for sites compared with MEP G1 and G2 sub-groups (reference sites are also represented to show the entire gradient). The clearest patterns were verified for similar pressure variables (TDS, turbidity, total nitrogen, total phosphorus, P-ortho, TSI1 and TSI2) (Figures 4 and 5).

Table 5. Attribution of test sites to their biological group (SIMPER) and quality class, according to 3- and 4-classes quality systems for all test sites.

Test sites	Group membership (group,%)	Quality class (3- classes system)	Quality class (4-classes system)
17	G1, 52.07	2	2
29	G1, 41.29	1	1
31	G2, 56.33	2	2
32	G2, 58.90	2	2
33	G1, 56.93	2	2
34	G2, 68.90	2	3
35	G2. 49.97	2	2
36	G1, 63.25	2	3
37	G2, 65.22	3	3
38	G2, 64.61	2	3
39	G2, 56.45	2	2
40	G2, 63.79	2	3
41	G1,69.49	2	3
42	G2, 69.96	3	3
43	G1, 74.70	3	4
44	G1, 67.32	3	3
45	G1, 77.42	3	4
46	G1, 70.30	3	3
47	G2, 61.55	2	3
48	G1, 68.61	3	3
49	G1, 68.84	3	3
50	G1, 75.70	3	4
51	G2, 57.09	2	2
52	G2, 52.42	2	2
53	G2, 57.36	2	3
54	G2, 60.49	2	3
55	G1, 74.08	3	3
56	G2, 79.70	3	4
57	G2, 56.14	2	2
58	G2, 56.15	2	2
59	G2, 71.17	2	3
60	G2, 58.98	2	3
61	G1, 72.66	3	3
62	G1, 68.97	3	3
63	G1, 66.03	2	3
64	G1, 77.55	3	4
65	G1, 74.85	3	4
66	G1, 82.35	3	4
67	G1, 68.74	3	3
68	G1, 80.72	3	4
69	G1, 90.80	3	4
70	G2, 60.16	2	3

Cont. Table 5.

71	G2, 59.40	2	3
72	G1, 72.58	3	3
73	G1,79.39	3	4
74	G1, 90.35	3	4
75	G1, 92.36	3	4
76	G1, 89.47	3	4
77	G1, 98.39	3	4
78	G2, 80.65	3	4
79	G2, 63.37	2	3
80	G1, 74.21	3	3
81	G2, 53.42	3	2
82	G1, 81.56	3	4
83	G1, 74.54	3	4
84	G2, 72.34	3	4
85	G2, 63.17	2	3
86	G1, 72.41	3	3
87	G1, 72.09	3	3
88	G1, 73.18	3	3
89	G2, 72.55	3	4
90	G2, 73.67	3	4

The first two canonical correlations axes showed a good strength for the association between the multivariate data in the Canonical Analysis of Principal Components (CAP) and the quality classes attributed by the 3-classes system for both G1 (δ_1= 67%, δ_2= 24%) and G2 (δ_1= 75%, δ_2= 10%) models.

For G1, the pressure variables TDS, chlorophyll *a*, total phosphorus (-0.75, -0.66, -0.62, respective), were those better correlated with the CAP axis 1 and the variable roads (0.65) with axis 2 (Fig. 6A). For G2, the pressure variables TDS, total nitrogen and commercial buildings (0.85, 0.75 and 0.78 respectively), were those better correlated with the CAP axis 1 and the variables odour and pasture (-0.65, 0.62) with axis 2 (Fig. 6B).

Regarding the 4-classes system, the first two canonical correlations axes showed also very good strength for the association between the multivariate data in the Canonical Analysis of Principal Components (CAP) and the quality classes for both G1 (δ_1= 84%, δ_2= 60%) and G2 (δ_1= 97%, δ_2= 48%) models. For G1 the pressure variables TSI2, TSI1 and odour (-0.72, - 0.62, -0.61, respectively), were those better correlated with the CAP axes 1 and the variable roads (0.61) with axis 2 (Fig. 7A). The pressure variables total nitrogen, TDS and commercial buildings (-0.92, 0.84, -0.84 and -0.75 respectively), were those better correlated with the CAP axes 1 and the pressure variable pasture (-0.62) with axis 2 (Fig. 7B).

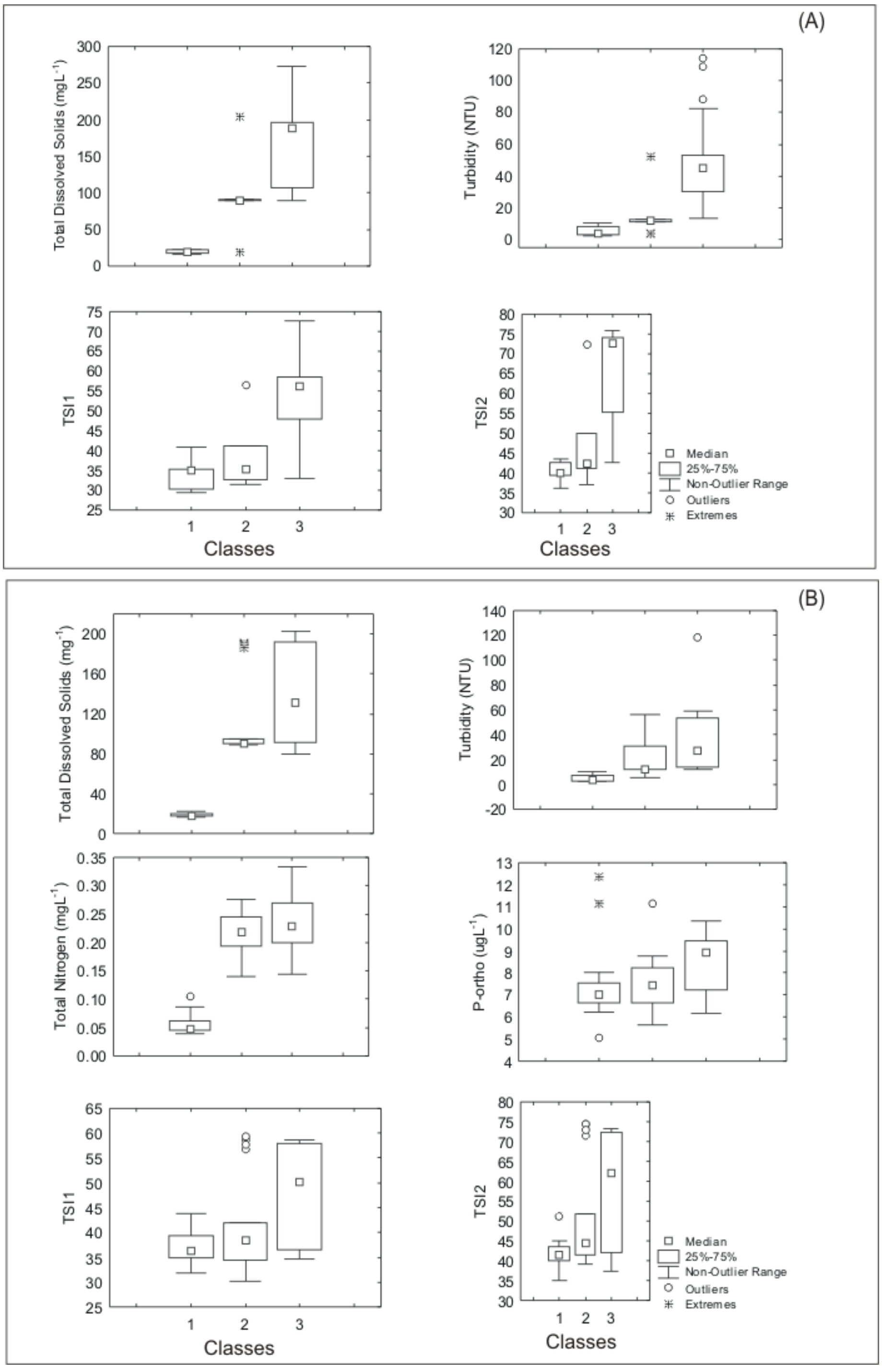

Fig. 4 Box-and Whisker plots of the disturbance variables against the classes attributed by the models based on MEP group G1 (A) and group G2 (B), for the 3-classes quality system. Outliers are 1.5 times outside the 25th and 75th percentile.

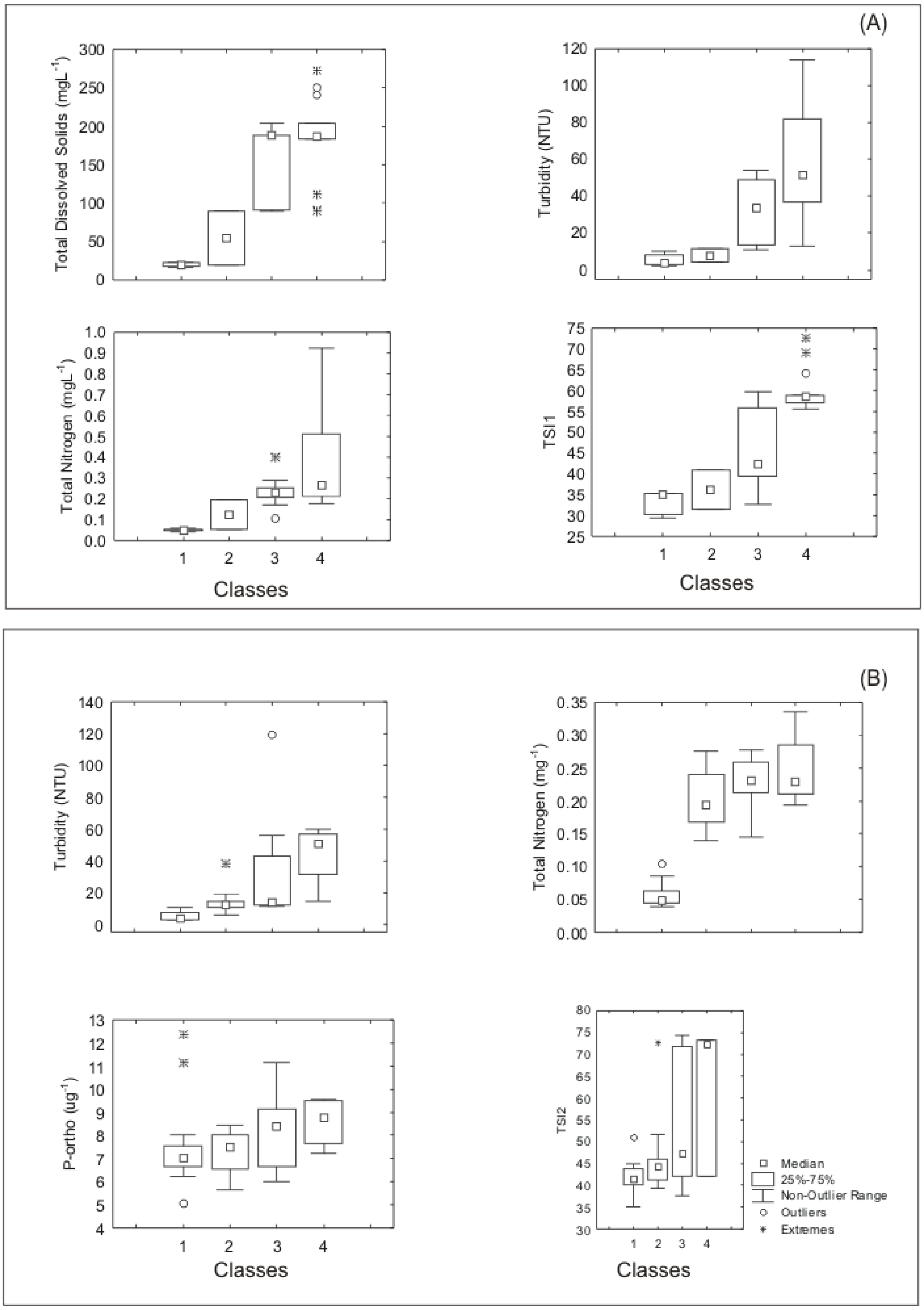

Fig. 5 Box-and Whisker plots of the disturbance variables against the classes attributed by the models based on MEP group G1 (A) and group G2 (B), for the 4-classes quality system. Outliers are 1.5 times outside the 25th and 75th percentile.

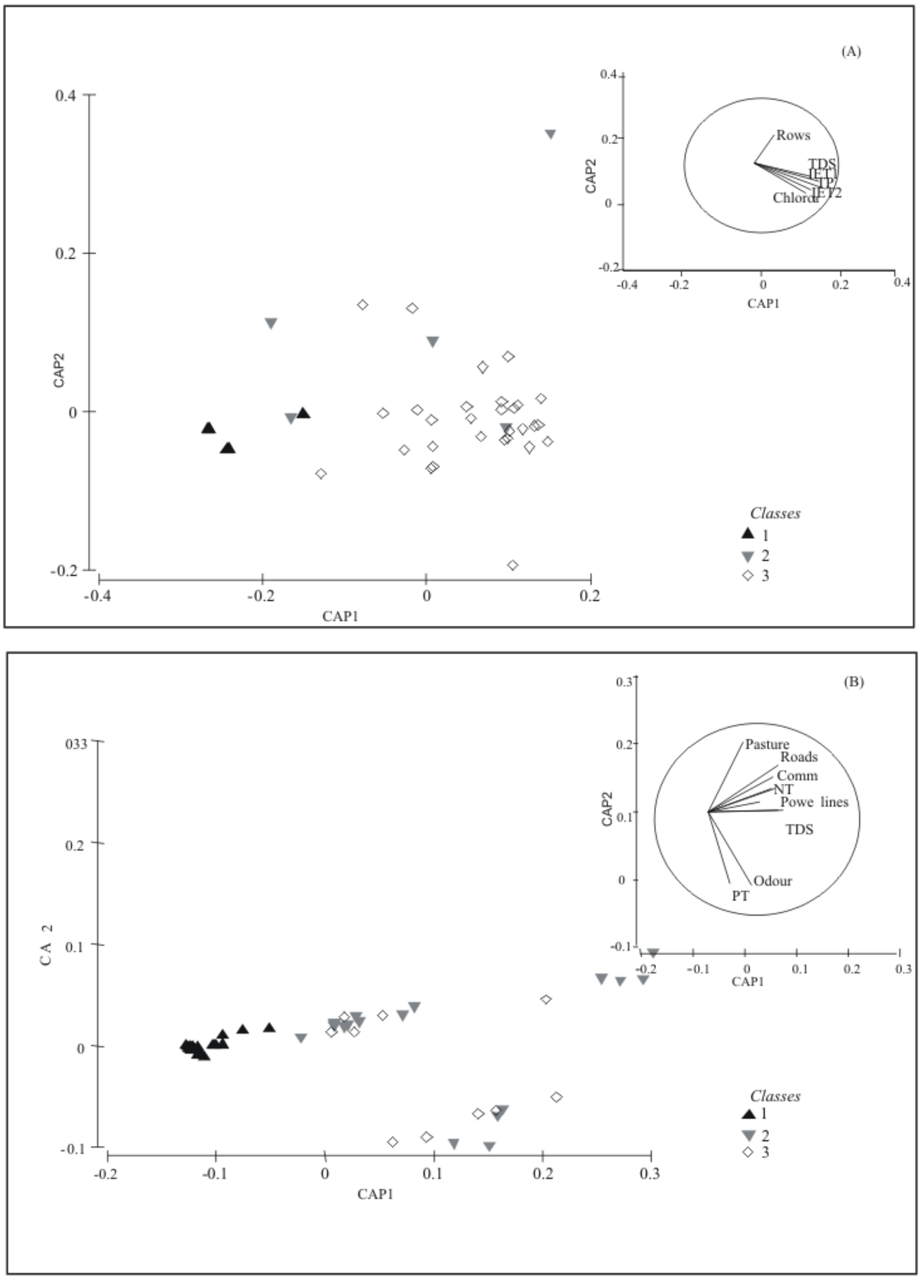

Fig. 6 Plots of Canonical Analysis of Principal Coordinates (CAP) based on disturbance data and with sites classified according to the 3 classes-system and for the models based on G1 (A) and G2 (B). Above are represented the vector overly of Spearman rank correlations of individual pressures variable vectors with the CAP axes (see Table 1).

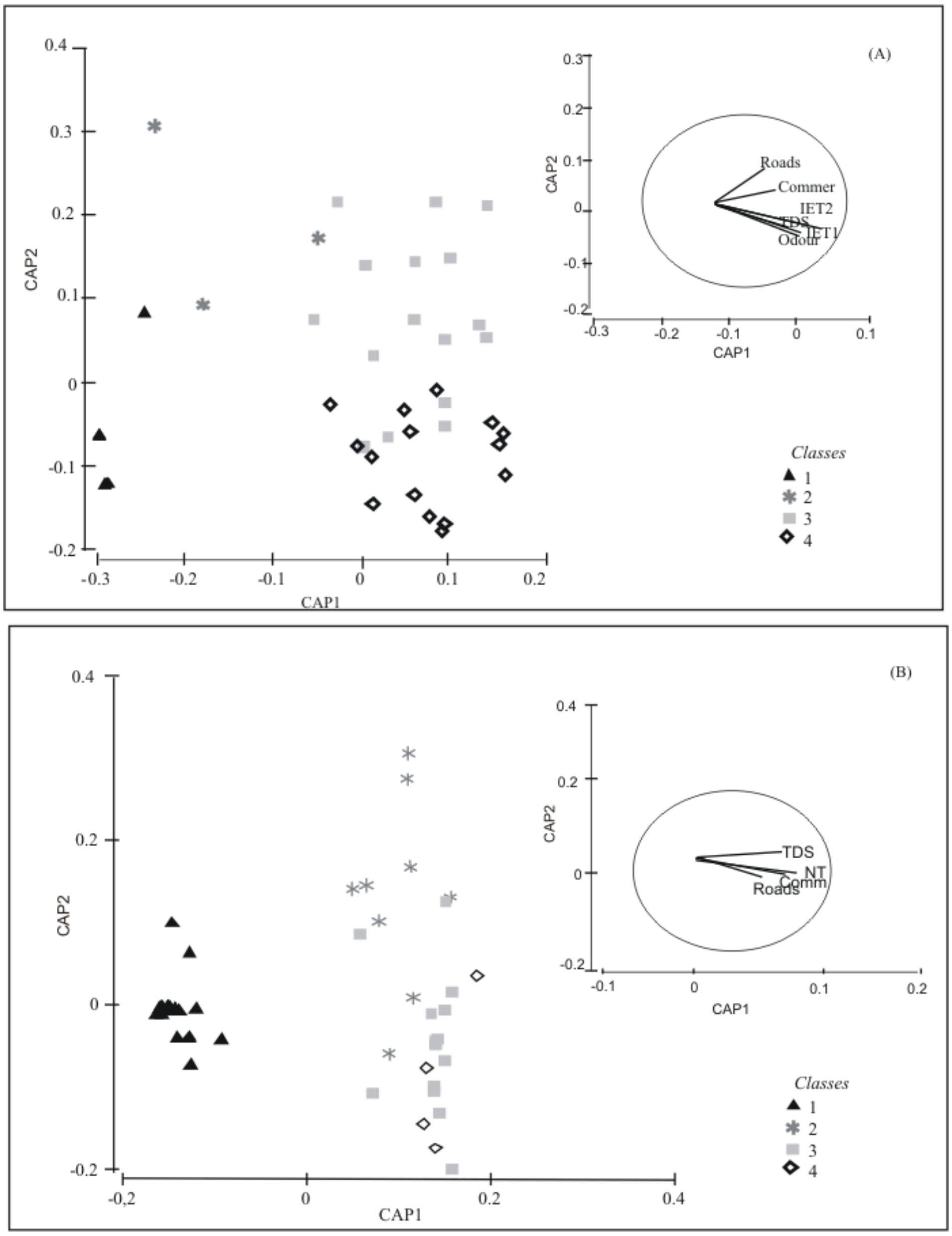

Fig. 7 Plots of Canonical Analysis of Principal Coordinates (CAP) based on disturbance data and with sites classified according to the 4 classes-system and for the models based on G1 (A) and G2 (B). Above are represented the vector overly of Spearman rank correlations of individual pressures variable vectors with the CAP axes (see Table 1).

Table 6. Results of PERMANOVA test for differences in pressure level between quality classes, for the two biological MEP (G1 and G2) and for the two classification systems.

	3 - Classes		4 - Classes	
	G1 (t, P(perm))	G2 (t, P(perm))	G1 (t, P(perm))	G2 (t, P(perm))
1-2	2.94; <0.001	2.27; <0.001	3.77; <0.001	2.80; <0.001
1-3	5.65; <0.001	2.28; <0.001	4.40; <0.001	3.58; <0.001
1-4			5.16; <0.001	4.15; <0.001
2-3	1.56; <0.05	1.37; <0.05	1.16; >0.05	1.30; >0.05
2-4			1.15; >0.05	1.66; >0.05
3-4			0.88; >0.05	0.86; >0.05

Discussion

Predictive models have been considered useful and accurate tools in the bioassessment of rivers for invertebrate communities (e.g. Norris & Norris, 1995; Reynoldson et al., 2001; Feio et al., 2007b; Hawkins et al., 2010), and also in fewer cases for lakes with different aquatic communities (Olden & Jackson, 2001). We showed here that this approach can also be used for reservoirs even though adjustments need to be made, such as the definition of “reference”, the nomenclature of quality classifications, the number of classes used, and the taxonomic level.

The maximum ecological conditions that we can presently find for reservoirs, to be used as “reference”, correspond to already poor (low richness) communities (Furey et al., 2006) which are very different from those that would be found in the previous system, i.e. the river, and in most cases it is not expectable to return to the original situation (Horsák et al., 2009). Therefore, the terms high, moderate, bad, even when talking about the ecological potential, as defined in the European WFD for the artificial and highly modified water bodies, did not seem appropriated. Alternatively, and following the approach used in the BEAST predictive model, developed for Great Lakes (Reynoldson et al., 1995; Reynoldson et al., 2000), we opted to define the degradation classes in terms of difference to MEP (ranging from Maximum Ecological Potential condition to a status Very Different from Maximum Ecological Potential).

Most of the water quality indices for lentic water bodies use 5 degradation classes, as the Lake Bioassessment Integrity Index (LBII) (Lewis et al., 2001), the Index of Size Distribution (ISD) (Reizopoulou & Nicolaidou, 2007) or the Pond Condition Index (PCI) (Trigal et al., 2007). However, the lack of agreement of the model classes with the gradient of impact would result in a non-functional model (Hawkins et al., 2000). Therefore, we opted to test which should be the most appropriate division in quality classes knowing that we can define the best possible invertebrate communities for the tropical reservoirs of the study area based on real data. We concluded that a classification system with only 3 quality classes (Equivalent to Maximum Ecological Potential, Different and Very Different from Maximum Ecological Potential) is enough and more effective in showing the anthropogenic degradation of the systems. This is not unexpected since the remaining gradient from a Maximum Ecological Potential community is already small (i.e. if we start with a community with few species the degradation needed to reach one or none species is much smaller than when the community is very diverse).

The accuracy of prediction models is obviously partially dependent on the number of reference groups established in the classification and in the power of the environmental descriptors in discriminating correctly the sites to its respective reference group. Many works point out that a model with a smaller number of groups means that each group has a higher intrinsic variability, limiting the sensitivity of the model (Feio et al., 2007b; Reynoldson et al., 1997; Reece et al., 2001). In our model we obtained a high accuracy (96%) of the environmental variables in predicting reference sites for the groups (Molozzi et al., *submitted*). Moreover, considering the relatively homogenous conditions found in the reservoirs, we consider appropriate the use of two groups.

The level of taxonomic resolution is also a key factor in defining which model should be used (Stribling et al., 2008). A high taxonomic level such as order or class may not be able to detect the different degrees of impacts while a lower taxonomic level such as species or genus can provide more information about the changes in the communities, but may lead to erroneous assessments due to mistakes made in identifying organisms to this taxonomic level (Feio et al., 2006; Stribling et al., 2008; Buss & Victorino, 2010). In our study we used mainly family level and in fact, we think that for our reservoirs a lower taxonomic level for the Ephemeroptera and Trichoptera groups is not relevant since only few individuals of this taxa were collected (Table 3). Opposite, we believe that improvements in models sensitivity, could be made if the Chironomids, that were already at genus level, were identified to species level as it is known that species of the genus found with high abundances (*Procladius*, *Chironomus*, *Tanytarsus*) present different sensitivities to environmental conditions and also to chemical, organic, and metal contaminants (Mousavi, 2002; Mousavi et al., 2003; Puntí et al., 2009).

However, in regular monitoring programs, the use of lower taxonomic levels would have the disadvantages of requiring a higher level of expertise from technicians and also more time to process the samples. Additionally, in the tropical areas, the knowledge of Chironomidae species (as for other groups) is still reduced. To avoid this some authors proposed alternative methodologies such as the use of pupal exuviae (Wilson & McGill, 1977; Wright et al., 1996; Callisto & Goulart., 2005; Ruse, 2010).

In conclusion, we think that our conceptual approach and the predictive model based on benthic macroinvertebrates communities can be useful in the bioassessment of the studied reservoirs and be the basis for monitoring schemes for other heavily modified water bodies. The division in 3 classes, from Equivalent to Maximum Ecological Potential to Very Different from Maximum Ecological Potential, instead of the traditional 4 or 5, traduced well the abiotic degradation. Nonetheless, this does not exclude future improvements, such as the increase of taxonomic resolution for Chironomids whenever the scientific knowledge allows it.

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CAPÍTULO 6

Potential of thermodynamic oriented ecological indicators as tools for environmental management in tropical reservoirs

Joseline Molozzi[1], Fuensanta Salas[3], João Carlos Marques[2]

& Marcos Callisto[1]

Reservatório de Ibirité

1- Laboratório de Ecologia de Bentos, Departamento de Biologia Geral, Instituto de Ciências Biológicas, ICB, Universidade Federal de Minas Gerais, Brasil
2- Intituto do MAR-CMA, Departamento de Ciências da Vida, Universidade de Coimbra, Portugal
3- TRAGSA, Murcia, Espanha

Submeter à Ecological Modelling

Potential of thermodynamic oriented ecological indicators as tools for environmental management in tropical reservoirs

Joseline Molozzi, Fuensanta Salas, João Carlos Marques & Marcos Callisto

Abstract: Approaches to evaluate ecological quality in aquatic ecosystems have become an important point among researcher and environmental managers worldwide. The potential of thermodynamic oriented ecological indicators as tools for environmental management, as compared to diversity indices, was tested in three tropical reservoirs located in the hydrographic basin of the Paraopeba River, Minas Gerais State-Brazil. We computed Eco-Exergy based indices, as well as the Margalef, and Shannon-Wiener indices, and tested differences across the reservoirs characterised by different anthropogenic disturbance. Indices were then compared with biotic (macrofauna biomass and abundance) and abiotic descriptors (pH, conductivity, transparency, turbidity, nutrients concentration, dissolved oxygen, chlorophyll *a*, and total dissolved solids). Results show that the ecological indicators selected were effective in detecting the ecosystem conditions. The Margalef index exhibited significant differences between the two areas with the reference sites presenting the highest values ($PseudoF_{2.719}$ = 24,506, p = 0.001). In the same way, the Shannon-Wiener values showed significant and higher values in the non perturbed areas ($PseudoF_{2.719}$ = 5.60, p = 0.042). The biomass pattern observed in our studied reservoirs leads us to consider that higher values of the Eco-exergy should be indicative of impacted sites. The index presented significantly higher values in the stations located in the disturbed reservoirs ($PseudoF_{2.719}$= 80.319; p = 0.001). Thermodynamic oriented indicators seemed to have provided useful information about the structural development of the community. However, the application of Eco-Exergy based indices requires further studies in tropical systems due to the unexpected biomass patterns observed in the present study, as well as in other studies carried out in similar geographical areas.

Keywords: Thermodynamic oriented indicators**,** Eco-exergy, diversity measures, benthic communities, monitoring, reservoirs.

Introduction

The reservoirs are highly modified ecosystems, built in response to the demands of economic growth. These systems represent a new insertion point of view of aquatic ecosystems, promoting significant changes in the hydrological and ecological distribution of rivers and watersheds (Tundisi, 2006). The instability of the new environment formed, resulting not only from the impact of impoundment, but also from perturbations induced by urbanization, agricultural and industrial activities, makes that the aquatic communities become unstable and gradually simpler (Fore et al., 1994; Klemm et al., 2003).

Large river–reservoir systems are some of the most difficult aquatic ecosystems to assess because they typically lack minimally disturbed reference sites, they are not natural systems to begin with, and reservoirs with high exchange rates are transitional systems between rivers and lakes (Terra & Araújo, 2011).

To evaluate the ecological status of the aquatic communities, a panoply of ecological indicators has been used. Nevertheless, in most cases, ecological indicators either only take into consideration some components of the ecosystem or result from non-universal theoretical approaches. In general terms, a number of them is based on the presence/absence of indicator species, other take into account the different ecological strategies carried out by organisms, as diversity, or the energy variation in the system through changes in species biomass. Another group of ecological indicators is either thermodynamically oriented or based on network analysis, looking to capture the information on the ecosystem from a more holistic perspective (Salas, 2002; Salas et al., 2006; Marques et al., 2009).

In general terms, the characteristics that would define a good indicator are easy handling sensibility to small variation of pollution, the type of specificity regarding pollution, independence of reference states, applicability in extensive geographical areas (Salas, 2002). Moreover, Salas et al. (2005) and Marques et al. (2009) considered that the excellent indicators are those based on the more general properties of populations, communities and on processes involved in ecosystem's functioning.

In this sense, Eco-exergy (Jørgensen & Mejer, 1979; Marques et al, 1997; 2003) is one of the mathematical functions that have been proposed as holistic ecological indicators in the last two decades, intending (a) to express emergent properties of ecosystems arising from self-organization processes in the run of their development and (b) to act as orientors (goal functions) in models development. Such proposals resulted from a wider application of theoretical concepts, following the assumption that it is possible to develop a theoretical framework able to explain ecological observations, rules and correlations on basis of an accepted pattern of ecosystem theories (Patricio et al., 2006).

Eco-exergy is a concept derived from thermodynamics that represents a measure of the maximum amount of work that the system can perform when it is brought into thermodynamic equilibrium with its environment. The Eco-exergy is a measure of the distance of the thermodynamic system from the equilibrium with the surrounding environmental, and therefore, it is the quantitative and qualitative measure of the energy. The Eco-exergy of an ecosystem at thermodynamic equilibrium would be zero. This means that, during ecological succession, Eco-exergy is used to build up biomass, which in turn stores Eco-exergy, and Eco-exergy therefore represents a measure of the biomass structure plus the information embedded in the biomass (Jørgense & Mejer, 1979; Jørgensen, 2002; Xu et al., 2005).

If the total biomass in the system remains constant then Eco-exergy variations will rely upon its structural complexity. Specific Eco-exergy is defined as Eco-exergy/biomass. Both Eco-exergy and Specific Eco-exergy may be used as indicators in environmental management. It might be advisable to use them complementarly.

Salas et al. (2005) stated that higher values of Eco-exergy and Specific Eco-exergy are indicators of higher biodiversity, higher functional redundancy, higher buffer capacity and resilience and more complex systems. That is the reason why the Eco-exergy and the Specific Eco-exergy have been used as indicators of the state of ecosystems in a number of European lakes (Jørgensen, 2000; Jørgensen et al., 1995; Nielsen, 1994; Ludovosi & Poleti, 2003; Jørgensen & Ulanowicz, 2009; Xu et al., 2005, 2011), coastal lagoons (Salas et al., 2005), freshwater systems and estuaries (Marques et al., 1997; Jørgensen & Padisak, 1996; Patricio et al., 2009) and coastal areas (Patricio et al., 2006; Salas et al., 2006).

The main aim of the present study is to test the ability of the Eco-exergy and Specific Eco-exergy to act as indicators of health state in three tropical reservoirs, ascertaining whether these indices are able to differentiate areas with maximum ecological potential (reference sites) from impacted sites based on benthic macroinvertebrates communities. We compared the estimations of Eco-Exergy based indices with the values of diversity indices (Shannon-Wiener and Margalef indices) to analyse the coherence of these different types of indicators in describing the state of the ecosystem.

Material and methods

Study site

In total, 3 reservoirs (Ibirité, Vargem das Flores and Serra Azul) were sampled in the Paraopeba river watershed, an affluent of the São Francisco River in the Minas Gerais state, south-eastern Brazil. The climate of this region is considered tropical sub-humid (Cwb), with summer rains (November to April) and a dry winter (May to October). The average annual temperature is ca. 20°C (Moreno & Callisto, 2006) (Fig. 1).

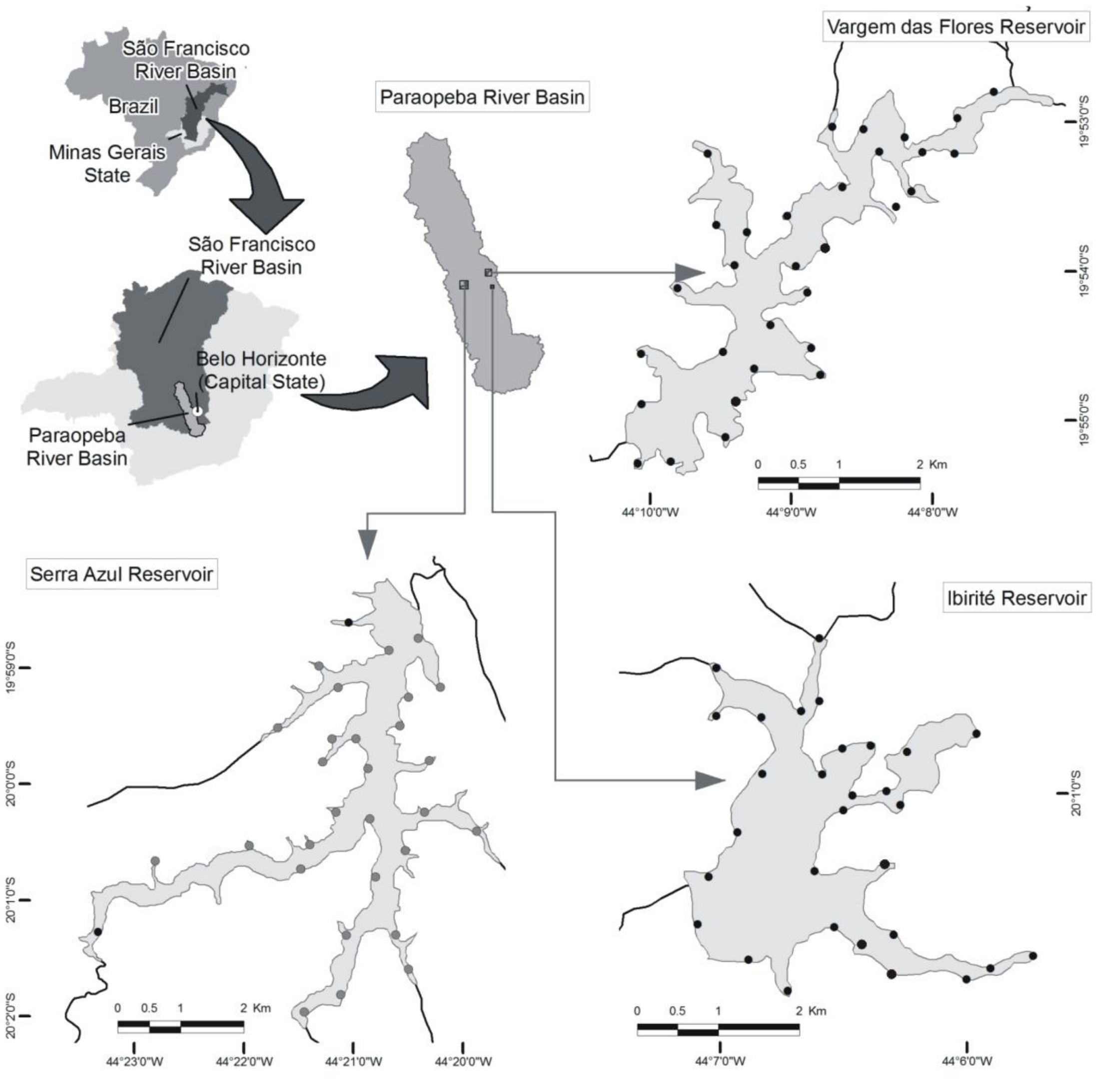

Fig. 1 Location of the reservoir Vargem das Flores, Ibirité and Serra Azul in the catchment of the Paraopeba River, Minas Gerais, Brazil and distribution of the sampling sites: Maximum Ecological Potential (reference sites) (●) and Impacted sites (●) in the reservoirs.

The Ibirité reservoir (20º01'13.39" S; 44º06'44.88" W), was built in 1968 at an altitude of 773 m a.s.l. This reservoir has an area of 2.8 km^2, a volume of 15,423,000 m^3 and an average depth of 16 m. The landscape of the reservoir basin is dominated by Eucalyptus plantations, a large condominium, small farms, and several industrial plants (Pinto-Coelho et al., 2010).

The Vargem das Flores reservoir (19º54'25.06" S – 44º09'17.78" W), was built in 1971 and is situated at 837 m a.s.l. It supplies water for the cities of Contagem and Belo Horizonte. The reservoir has a surface area of 4.9 km^2, contains 37,000,000 m^3 of water and has a maximum depth of 18 m. The maximum height of sill spillway is 837 m and the reservoir as a hydraulic retention time of 365 days. However, the population around the reservoir reaches about 100,000 people, and was transformed into an area leisure for the region (COPASA, 2004).

The Serra Azul reservoir (19º59'24.92" S; 44º20'46.74" W) was built in 1981, is located at an altitude of 760 m a.s.l., and has a water surface of 7.5 km^2, water volume of 88,000,000 m^3 and a maximum depth of 40 m. The maximum height of sill spillway is 760 m and the reservoir has a hydraulic retention time of 351 days. Surrounding this reservoir there is also a special protected area established in 1980 with an area of 27.000 ha. Inside this area, 3.2 ha belong to the COPASA, the industry that explores the reservoir (COPASA, 1980), and no tourism or fisheries are allowed.

Sampling stations

In total, 90 sampling sites were sampled in the 3 reservoirs at sites representative of the different biocenosis, reference sites and main polluted areas. The samples stations in the reservoirs were sampled quarterly - March, June, September and December in 2008 and 2009.

Twenty eight different stations located in the Serra Azul reservoir were classified in a previous work (Molozzi et al., submitted) with Maximum Ecological Potential (MEP). Therefore, these 28 stations were considered in the present study as reference sites.

Environmental parameters

At each sampling occasion the following physical and chemical parameters of water were measured using an YSI Model Multiprobe: dissolved oxygen, conductivity, turbidity, total dissolved solids (TDS) and pH. Additionally, sub-surface water samples were collected with a Van Dorn type cylinder for subsequent measurement of total nitrogen (TN), total phosphorus (TP) and orthophosphate (PO_4), in accordance with "Standard Methods for the Examination of Water and Wastewater" (APHA, 1992). The concentration of chlorophyll *a* (Chl*a*) was obtained according to Golterman et al. (1978). The depth of the euphotic zone (S) was determined based on the readings of the Secchi disc. The depth of the water column was estimated using a portable sonar.

Biological samples

Macroinvertebrates were collected with an Eckman-Birge dredge (0.0225 m^2), as close as possible to the margin and at a median depth of 3.92 m. The collected material was fixed with 70% formalin and was transported to the laboratory. Invertebrates were mostly identified to the family level (Merritt & Cummins, 1996; Fernandez & Dominguez, 2001; Costa et al., 2006; Mugnai et al., 2010). Chironomidae larvae were identified to genus, treated with 10% solution of lactophenol and identified under a microscope (400x) with the aid of Trivinho-Strixino (2011) and Epler (2001) identification keys. After the identification and determination of taxonomic composition, the organisms were dried in an oven at 60 ° C for 48 hours and weighed with a precision of 0.01 mg to determine their biomass (gm^{-2}). Mollusca were burned in furnace at 500° C for 4 hours in order to estimate the weight of the mineral fraction. The biomass of this group was calculated by the difference between the weight of the containers with and without organisms discounting the value of the mineral part (Elkarmi & Ismail, 2007).

Computation of the indices

Eco-exergy estimations

If Eco-exergy is calculated only from the chemical potentials, which are extremely dominant with regard to ecosystems, the expression given in equation 1 is valid with good approximation (Jørgensen, 2002). Detritus was used as reference level, i.e., $\beta i = 1$ and Eco-exergy in biomass of different types of organisms is expressed in detritus energy equivalents (eq. 1). This formulation does not correspond to the strict thermodynamic definition, but provides nevertheless an approximation of Exergy values. In this sense it was proposed to call it Eco-exergy index (Marques et al., 1997).

$$Eco\text{-}exergy = T \times \Sigma \; C_i \times \beta_i \qquad (eq.\ 1)$$

Where T is the absolute temperature, C_i is the concentration for component i in the ecosystem (e.g. biomass of give taxonomic groups), β_i- is a factor able to roughly the quantity of information embedded in the genome of the organisms.

The important stage of Eco-exergy calculation is the assessment of the conversion factor (β_i). The factor is determined by the degree of give species organism complexity, depending on is evolutionary development and calculated on the number of informative genes and number of cell kinds of given organism. β values have previously been calculated for several organisms based upon number of coding genes (see Jørgensen et al., 2005 - Table 1).

If the total biomass in the system remains constant though time, then the variation of Eco-exergy will be a function of only the structural complexity of the biomass or, in other words, of the information embedded in the biomass, which may be called Specific Eco-exergy (SpEx) per unit of biomass (eq. 2).

Specific Eco-exergy is given by,

Specific Eco-Exergy = Eco-Exergy/Total Biomass *(eq. 2)*

Diversity estimations

We chose to use the Shannon-Wiener and Margalef indices to compare the values with estimations of the Eco-exergy index, to analyse the coherence of these different types of indicators in describing the state of the ecosystem.

Table 1. Exergy/Biomass conversion factors (β) for benthic communities, based in: Jørgensen et al., 2005.

Organism	**Exergy conversion factor (β)**	**Organism**	**Exergy conversion factor (β)**
Virus	1.01	Kinorhynch	165
Minimal cell	5.8	Gastrotric, MetI	76
Bacteria	8.5	Rotifera	163
Algae	20	Gnahostom	143
Archaea	13.8	Gastrotric, MetII	116
Protists	21	Ctenophora	167
Diatoms	66	Entoprocta	165
Yeast	17.4	Nematoda (Worms)	133
Fungi	61	Nematina	76
Protozoa, Amoebe	31-46	Mollusc	310
Prolifera	97	Gastropods	312
Angiosperm	147	Bivalve	297
Rhodophyta	92	Annelida (f.i.leeches)	133
Bryophyta	173	Brachiopods	109
Pteridophyta	146	Sea squirt	191
Psilophyta	170	Crustacean	232
Pinus mono	314	Coleoptera (Beetles)	156
Mustard weed	147	Diptera (Flies)	184
Rice	275	Hemiptera	159
Eudicot	268	Hymenoptera	267
Monocot	393	Lepidoptera	221
Placozoa	35	Phasmida	43
Cnidaria	91	Mosquito	322
Platyhelminthes	120	Chordata	246
Mesozoa	30	Fish	499

Shannon-Wiener (Shannon & Weaver, 1963) index is based on information theory and assumes that individuals are samples at random, of an "indefinitely large" community, and that all the species are represented in the samples and can be estimated according to the algorithm (eq. 3):

$$H' = - \Sigma\, p_i \log_2 p_i \qquad (eq.3)$$

where p_i is the proportion of the individuals belonging to species *i* in the samples. The real value of p_i is unknown, but is estimated through the ratio N_i / N, were N_i is the number of individual of the species *i* and N is the total number of individuals. The units for the index depend on the log used. So, for $\log_2$ the unit is bits/individual, 'natural bels' and 'nat' for log e.

The Margalef index (Margalef, 1969) quantifies the diversity relating specific richness to the total number of individuals (eq. 4).

$$D = (S-1)/\log_2 N \qquad (eq.4)$$

where *S* is the number of species and *N is* total number of individuals.

Data analysis

A previous work (Molozzi et al., submitted) carried out in the same study reservoirs showed that there is no pattern of high correlation between the communities sampled in the same month of the year (e.g. December 2008, December 2009) or the same season (dry, wet). Therefore, there was no reason for considering values of the structural parameters of the communities for different seasons, and the mean taxa abundance and biomass were used for each sampling site (Table 2).

In order to examine the similarity between communities from reference (28 sites, maximum ecological potential- Serra Azul reservoir) and impacted sites (62 sites in Serra Azul, Ibirité and Vargem das Flores reservoirs) multivariate analysis was performed using the PRIMER 6 software (Software package from Plymouth Marine Laboratory, UK). Data (species abundance and biomass) were transformed by square root. Bray-

Curtis similarity matrix was calculated and used to generate 2-dimensional plot with the non-metric multidimensional scaling (nMDS) technique (Clarke & Warwick, 2001; Clarke & Gorley, 2006). Stress values were shown for each nMDS plot to indicate the goodness of representation of differences among samples. An ANOSIM was applied to see which of the proposed groups were significantly distinct.

Table 2. Results of ANOSIM pair wise tests between the samples of Serra Azul, Vargem das Flores, and Ibirité reservoirs; ns indicates p values >0.05.

Months	**Serra Azul (R, p)**	**Vargem das Flores (R, p)**	**Ibirité (R, p)**
March/08 - March/09	0.08, 0.0004	0.18, 0.001	0.21, 0.001
March/08 - June/08	0.04, ns	-0.01, ns	0.04, 0.04
March/08 - June/09	0.09, 0.002	0.04, 0.05	0.07, 0.007
March/08 - September/08	0.02, ns	0.03, ns	0.09, 0.001
March/08 - September/09	0.09, 0.001	0.06, 0.015	0.39, 0.001
March/08 - December/08	0.04, 0.025	0.06, 0.015	0.21, 0.001
March/08 - December/09	0.15, 0.001	0.10, 0.001	0.31, 0.001
March/09 - June/08	0.05, 0.018	0.20, 0.001	0.20, 0.001
March/09 - June/09	0.02, 0.094	0.22, 0.001	0.14, 0,001
March/09 - Setember/08	0.05, 0.025	0.13, 0.003	0.10, 0.001
March/09 - Setember/09	-0.01, ns	0.25, 0.001	0.13, 0.005
March/09 - December/08	0.08, 0.004	0.14, 0.001	0.07, 0.005
March/09 - December/09	0.02, ns	0.34, 0.001	0.20, 0.001
June/08 - June/09	0.06, 0.015	0.1, 0.001	0.07, 0.007
June/08 - September/08	0.05, 0.023	0.01, ns	0.05, 0.03
June/08 - September/09	0.04, 0.035	0.15, 0.001	0.22, 0.001
June/08 - December/08	0.03, 0.038	0.07, 0.015	0.21, 0.001
June/08 - December/09	0.10, 0.002	0.21, 0.001	0.26, 0.001
June/09 - September/08	0.04, 0.029	0.09, 0.004	0.11, 0.001
June/09 - September/09	0.03, 0.054	-0.02, ns	0.17, 0.001
June/09 - December/08	0.08, 0.002	0.12, 0.001	0.25, 0.001
June/09 - December/09	0.04, 0.036	0.01, ns	0.22, 0.001
September/08 - September/09	0.05, 0.014	0.13, 0.004	0.11, 0.003
September/08 - December/08	0.01, ns	0.02, ns	0.09, 0.006
September/08 - December/09	0.10, 0.001	0.18, 0.001	0.23, 0.001
September/09 - December/08	0.06, 0.008	0.16, 0.001	0.18, 0.001
September/09 - December/09	-0.01, ns	0.001, ns	0.06, 0.007
December/08 - December/09	0.10, 0.001	0.16, 0.001	0.35, 0.001

Significant differences between reference and impacted sites with respect to Margalef index, Shannon-Wiener index, Eco-exergy and Specific Eco-exergy were evaluated with a PERMANOVA test with 999 permutations (Permutational Multivariate Analysis of Variance; Anderson, 2001a,b; Anderson & Braak, 2003, Anderson et al., 2008; software package PERMANOVA + for PRIMER, 2006). This routine is a multivariate permutational non-parametric test, analogue of the univariate ANOVA and graphically presented by Box-and-Whisker plots (Software Statistic 7.0).

Pearson's correlation ($p \leq 0.05$) were estimated to evaluate the relationships between the values of the indices and environmental factors (Software Statistic 7.0). The correlation coefficient itself, rather than its probability, is more critical as far as measuring agreement between parameters is concerned, because the coefficient reflects the ratio of covariance between different variables (Willby & Birk, 2010). Thus, a minimum value for r of 0.4-0.5 is required to consider a significant correlation between the tested indices and the environmental variables.

Results

Benthic communities response to pressures

Figure 2 illustrates the results obtained from the nMDS multivariate analysis for biomass and abundance data on the biological communities. The stress values obtained correspond to an useful two-dimensional representation, however the stress values are 0.19 and 0.20 respectively, after which the reliability of the detail of the graphic representations should be examined carefully.

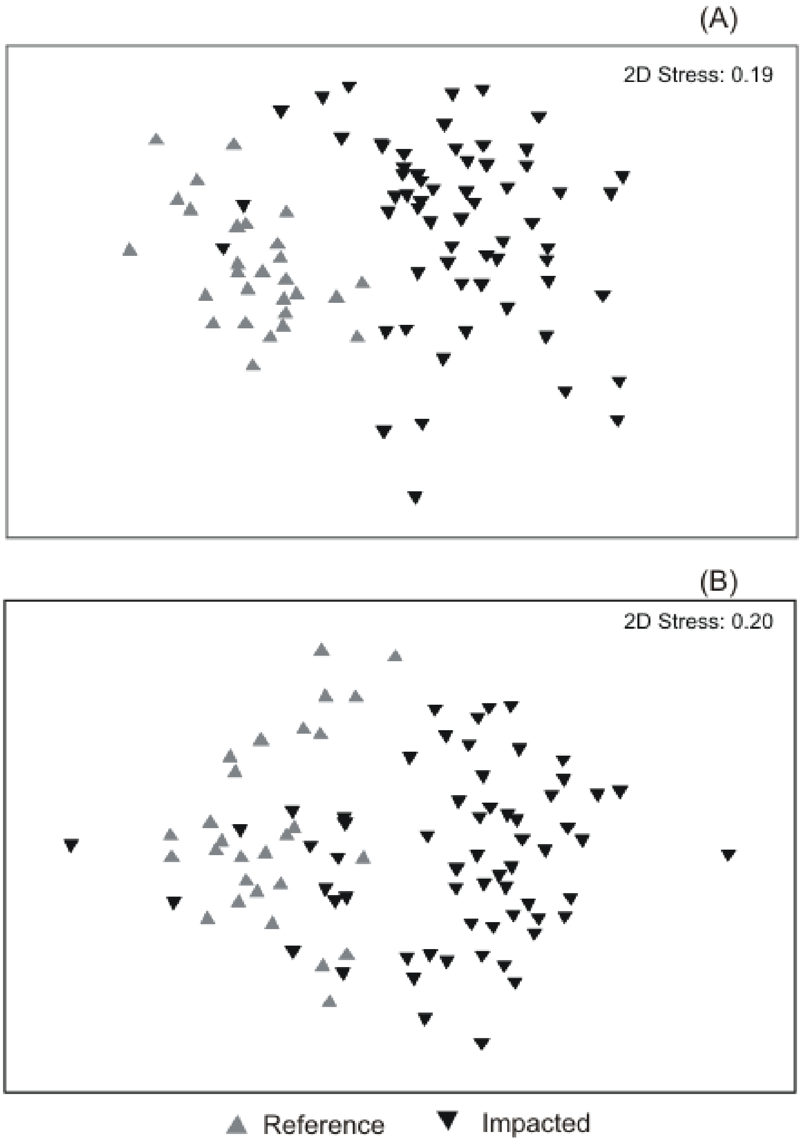

Fig. 2 nMDS ordination plot based on macrobenthic biomass data (A) and abundance data (B) from 90 sites located in the studied reservoirs.

ANOSIM analyses show significant differences (ANOSIM Biomass: Global R = 0.348, p = 0.001; ANOSIM Abundance: Global R = 0.463, p = 0.001) between the reference sites and impacted sites. Taxa such as *Melanoides tuberculatus*, Oligochaeta and *Chironomus*, were found in higher proportions in the impacted sites (25.47%, 24.83% and 9.27%, respectively) and in lower proportions in sites with Maximum Ecological Potential (2.62%, 3.42%, 1.05%, respectively). The taxa *Fissimentum*, Philopotamidae, Hydrobiosidae and *Procladius* were found in high proportions in sites with Maximum Ecological Potential (5.45%, 0.04%, 0.04% and 3.16% respectively) and with low percentages or absent in impacted sites (0.24%, 0%, 0%, 0.09%, respectively).

Regarding biomass, in both sampling years the highest values for all taxonomic groups were obtained in impacted sites (Fig. 3).

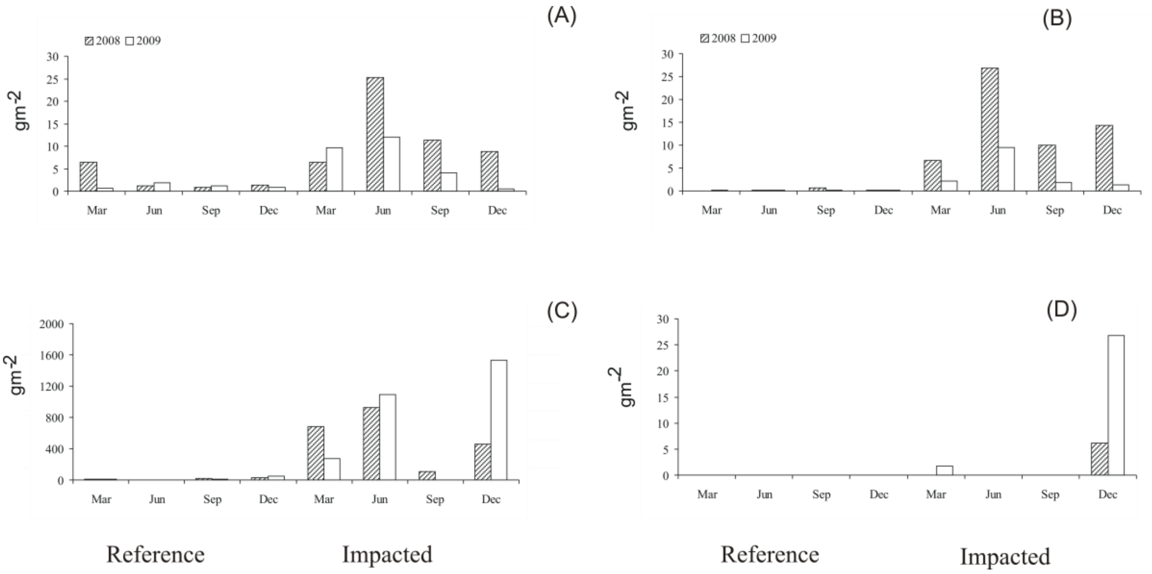

Fig. 3 Temporal and spatial variation from 2008 and 2009, the biomass of the groups used the calculated exergy index the sites reference and sites impacted: (A) Diptera, (B) Annelida, (C) Gastropoda and (D). Bivalve

The taxonomic group contributing the most to the total biomass was Gastropoda due to the presence of the exotic species *Melanoides tuberculatus.* The total biomass of *M. tuberculatus* represented 5,484.55 gm^{-2} in impacted sites, and 140.20 gm^{-2} in sites of reference. The biomass of Annelida (Oligochaeta and Hirudinea) was also much higher in the impacted sites than in reference sites (63.96 gm^{-2} and 1.11 gm^{-2} respectively), and the same pattern was observed for Diptera.

The high values of benthic biomass found in the disturbed sites have been also observed in the littoral zones of other tropical reservoirs in Brazil, and other authors (Takahashi et al., 2008) consider that this elevated biomass in eutrophic reservoirs may be related to a high productivity due to the elevated content of organic matter in the environment.

Indices performance

How do the different ecological indicators separate impacted and reference sites? Table 3 and Figure 4 show the results.

Table 3. Average values of Margalef, Shannon-Whiner, Eco-exergy and Specific Eco-exergy indeces in reference and impacted sites in three reservoirs in Brasil.

Index	**Reference sites**	**Impacted sites**
Margalef	1.19	0.8
Shannon-Wiener (Bits/ind)	0.87	0.85
Eco-exergy (gm^2 det energy equiv)	205.99	4,157.88
Specific Eco-exergy	291.23	272.43

The Margalef index (Fig. 4a) exhibited significant differences between the two types of sites with the references sites presenting the highest values ($PseudoF_{2.719}$ = 24.506, p = 0.001). In the same way, the Shannon-Wiener values (Fig. 4b) showed significant and higher values in the non perturbed areas ($PseudoF_{2.719}$ = 5.60, p = 0.042).

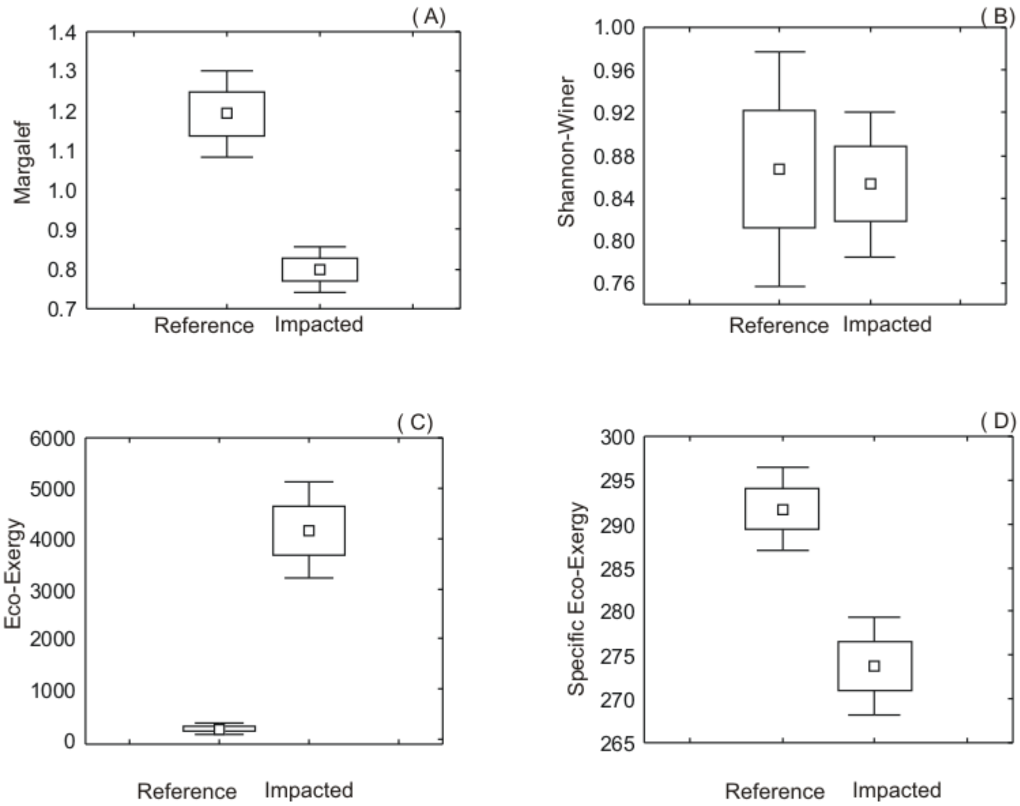

Fig. 4 Variation of the applied indices in the reference and impacted sites over two years. Margalef index (A), Shannon-Wiener index (B), Eco-exergy index (C) and Especifc Eco-exergy (D).

According to the hypothesis tested by Marques et al. (1997) one should expect that the Eco-exergy index would be able to reflect disturbance effects, with the lowest values being found in the impacted sites. However, the biomass patterns observed in our reservoirs and in other tropical reservoirs, lead us to consider that in our study higher values of the Eco-exergy should be indicative of impacted sites. In fact, the index presented significantly higher values in the stations located in the disturbed reservoirs ($PseudoF_{2.719}$= 80.319; p= 0.001) (Fig. 4c, Table 3).

Finally, the Specific Eco-exergy index, that quantifies the information quality embedded in the biomass, presented, as it was expected, higher values in the non disturbed areas, although the difference was non significant (Fig. 4d).

The analysis of the relationship between two different ecological indices may help understanding if both indices are measuring the same aspect of the community and in the same direction. Because the particular situation shown in this study, obtaining higher values of Eco-exergy in impacted sites due to the elevated biomass found in disturbed tropical reservoirs, we consider that a negative relation between the Eco-exergy and the other indices applied means a similar behaviour in response to the disturbance.

Considering the link among ecological indicators and environmental variables, Eco-exergy, Specific Eco-exergy, and Margalef were sensitive to several parameters usually associated with an impact.

The Margalef index showed significant negative correations with conductivity and TDS, usually associated to poor quality waters (Table 4).
Eco-Exergy showed a significant positive correlation with conductivity and pH, Specific Eco-Exergy showed positively correlated with water transparency (measured with secchi disc) and depth, and negatively correlated with conductivity and TN.

Discussion

Our results showed that the Eco-exergy index was able to separate the reference from the impacted sites and was able to clearly show the differences that occur within the communities. However, it is expected that reference sites present relatively higher Eco-exergy values when compared with impacted sites (Jørgensen, 1995; Marques et al., 1997, 2003; Salas et al., 2005).

Table 4. Values obtained after the application of Person's correlation between the different indices and the different environmental variables sampled in reservoir, Brazil. (*): r>0.4.

Variables	Margalef	Shannon	Eco-Exergy	Specific Eco-Exergy
Margalef	1.000			
Shannon-Wiener	0.673	1.000		
Eco-exergy	-0.280	-0.036	1.000	
Specific eco-exergy	0.115	-0.140	-0.116	1.000
Depht (m)	0.125	-0.137	-0.334	0.177
Secchi (m)	0.386	-0.134	-0.373	0.425*
pH	-0.360	-0.076	0.458*	-0.077
Conductivity (μScm^{-1})	-0.443*	0.090	0.466*	-0.441*
TDS (mgL^{-1})	-0.525	-0.005	0.379	-0.267
Turbidity	-0.331	0.013	0.248	-0.369
O_2 ($m\ L^{-1}$)	0.389	0.107	0.001	-0.014
Chlorophyll a ($\mu g\ L^{-1}$)	-0.322	-0.014	0.300	-0.120
Total nitrogen ($mg\ L^{-1}$)	-0.340	0.123	0.374	-0.446*
Total phosphorus ($\mu g\ L^{-1}$)	-0.187	0.104	0.057	-0.293
Orthophosphates ($\mu g\ L^{-1}$)	-0.054	0.099	-0.064	-0.223

According to Odum (1988) ecosystems responding to environmental stress, show the reduction in food chains, present reduced species diversity, an increase in the proportion of "r-strategists" species and reduced organism size to increase the efficiency of resource use. Our results showed that in response to stress, a reduction in diversity of species in the impacted sites occurred, as well as an increase of "r-strategists" species, however it was not observed a reduction in organism size (biomass). Far from it, and increase in biomass was observed in the disturbance areas, leading to the high values of Eco-exergy index recorded.

Studies carried out by Takahashi et al. (2008) in Brazilian reservoirs and Callisto et al. (2002) in lakes, showed the same biomass pattern observed in the present study. As in our case study, they found high values of Chironomidae biomass in impacted sites. Moreover, it should be emphasized that we found a higher abundance and biomass of *Chironomus* in the most impacted reservoir. Jorcin & Nogueira (2008) pointed out that this genera is tolerant to different climatic, hydrological, and limnological conditions, being associated with eutrophic environments.

But, what is the reason of increased biomass in impacted sites? Vos et al. (2000) showed that detritus with high nitrogen, phosphorous, carbon and fatty acid are associated with the presence of Chironomidae larvae with larger sizes. Although, it is not possible to discuss these particular relationships in the present study, it can be observed that detritus can be a determining factor in the composition, abundance and biomass of Chironomidae. Another factor that may be related to increased biomass of these organisms is that they are associated with organic matter rich sediment (fine particles, mud) used in the construction tubes, and that can also function as a shelter to avoid predators, because these organisms body colour (red) is easily visible (Butler & Anderson, 1990; Helson et al., 2006). Therefore, it is possible that the larvae that live in such shelter environments, even in the same stage, could present a greater biomass accumulation (Takahashi et al., 2008).

Another factor that may have contributed to the increase in the eco-exergy value of impacted sites may be associated with the high biomass of *Melanoides tuberculatus*. This species is an "r-strategist", with parthenogenetic reproduction and potential to maintain high population densities for long periods of time (Silva et al., 2010). These organisms are also highly adaptable and easily transported, establishing on all kinds of substrates (Livishts et al., 1983).

The *M. tuberculatus* has a sharp growth in the juvenile phase, resulting from increased investment to reach sexual maturity (which can be achieved within 90 to 279 days), reducing further growth and investing in reproduction thereafter (Dudgeon, 1986; Livshites & Fishelson, 1983; Elkarmi & Ismail, 2007). This organism has the greatest values of growth rate (2.83 $Kyear^{-1}$) than any other Gastropoda (Pointier et al., 1993).

This confirms the potential of this species to be classified as an "r strategist". In adverse conditions (low temperatures or lack of food) they can slow their growth or even hibernate for long periods or aestivation (Livshites & Fishelson, 1983; Supian & Ikhwanuddin, 2002), and this fact could explain the low biomass of these organisms in oligotrophic environments.

Moreover, such dominance of *M. tubercaluts* in eutrophic conditions could be responsible for the non detection of significant differences between compared reference and impacted values of Specific Eco-exergy. Each organism is represented by a conversion factor used in the calculation of Eco-exergy and Specific Eco-exergy. This factor is determined by the degree of given species organism complexity, depending on its evolutionary development and calculated based on the number of informative genes and the number of cell types of a given organism (see Table 1) (Jørgensen et al., 2005). Specific Eco-exergy, expresses the ability of an ecosystem to utilize available resources (Xu et al., 2001) and is obtained by dividing Eco-exergy by the total biomass, so when we remove the biomass we are left with the structural values of each organization Gastropods (mainly represented by *M. tuberculatus* in the present study) present a high conversion factor (β) (Table 1), and this fact has affected the Specific Eco-exergy values considerably.

However, Specific Eco-exegy seemed to have provided useful information about the environmental conditions, showing significant relations with indicator variables of poor quality water as conductivity, total nitrogen and transparency.

Due to the biomass patterns found in our tropical reservoirs and the dominance of the exotic species *M. tuberculatus*, it has not been possible to find a strong correlation between high Eco-exergy and high Specific Eco-exergy, as it should be expected as high

Eco-exergy is found for mature ecosystems that generally should have more developed organisms (Jørgensen, 2007).

Regarding diversity, the Margalef and Shannon-Wiener indices were able to distinguish different eutrophication levels. However, Shannon-Wiener index was not related to environmental parameters linked to disturbances, leading us to consider the Margalef index as a more sensitive indicator of eutrophication.

In general, the ecological indicators selected were effective in detecting the prevailing ecosystem conditions. Thermodynamic oriented ecological indicators seemed to have provided useful information about the structural development of the community. However, the Eco-exergy indices requires further studies in tropical systems due to the biomass patterns found in the present work and in other studies carried out in other tropical areas.

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CONSIDERAÇÕES FINAIS

- Além de variáveis físicas e químicas o grau de preservação do entorno de reservatórios também deve ser considerado como um fator estruturador de comunidades bentônicas na região litorânea de reservatórios.
- O uso integrado de variáveis abióticas, do protocolo de diversidade de habitas e das comunidades bentônicas, permitiu a seleção de locais com máximo potencial ecológico para os reservatórios na bacia hidrográfica do Rio Paraopeba (MG). Esta abordagem poderá ser um primeiro passo na seleção de locais de referência em reservatórios urbanos no Estado de Minas Gerais.
- Nos reservatórios estudados, os principais fatores tipológicos que influenciam a distribuição de comunidades bentônicas na região litorânea são relacionados com o tipo de substrato de fundo, o tipo de substrato na região litorânea e à profundidade.
- O biomonitoramento de comunidades de macroinvertebrados bentônicos na região litorânea de reservatórios permitiu propor e testar um modelo preditivo e índices para avaliar o gradiente de impactos a que os locais estão submetidos nos reservatórios.
- Através dos resultados desta tese reforçamos a importância de utilizar os macroivertebrados bentônicos na região litorânea de reservatórios urbanos na avaliação de qualidade ecológica.

Diante do exposto a conclusão final desta tese é de que a estrutura e distribuição de comunidades bentônicas na região litorânea de reservatórios na bacia hidrográfica do Rio Paraopeba (MG) são influenciadas por ações antrópicas na área de entorno. Além disso, os diferentes níveis de impactos a que estes reservatórios estão submetidos refletem um gradiente de degradação ambiental distinto para as comunidades bentônicas, corroborando a hipótese testada.

Assim, após os dois anos de amostragens, da utilização de diferentes ferramentas para avaliar a qualidade ecológica dos reservatórios, a conclusão apresentada é de que: 28 locais foram classificados no reservatório de Serra Azul com Máximo Potencial Ecológico (MEP), 2 locais foram classificados como Diferente do Máximo Potencial Ecológico. Para o reservatório de Vargem das Flores, 20 locais foram classificados como Diferente do MEP e 11 locais como Muito Diferente do MEP. No reservatório de Ibirité 5 locais foram classificados como Diferente do MEP e 24 locais Muito Diferente do MEP (Fig. 3). Essas informações devem ser analisas de forma cuidadosa pelos órgãos gestores desses reservatórios, a fim de gerirem programas de gestão e conservação dessas massas de água altamente modificadas.

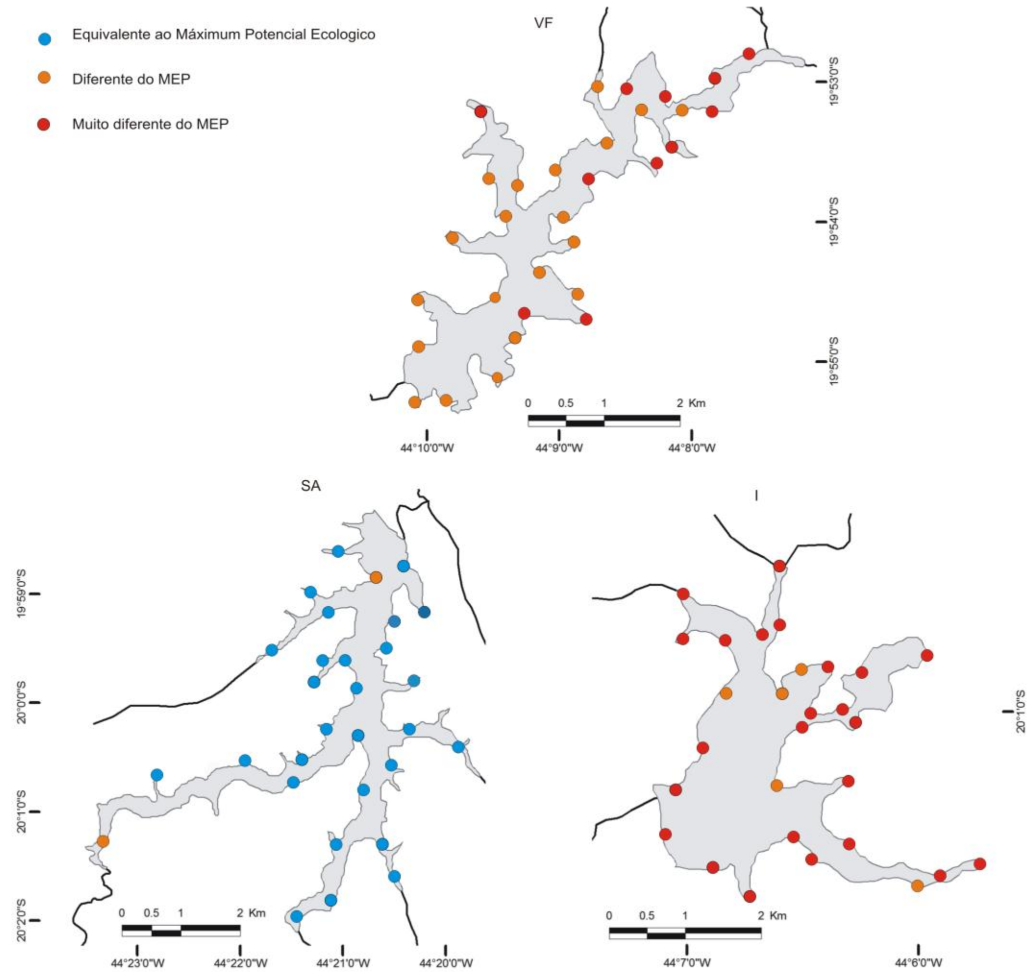

Fig. 3 Classificação final da qualidade ecológica dos reservatórios de Vargem das Flores (VF), Serra Azul (AS) e Ibirité (I), na bacia hidrográfica do Rio Paraopeba (MG), após dois anos de amostragens.

PERSPECTIVAS FUTURAS

Os resultados alcançados nesta tese nos incentivam a acreditar que as diferentes metodologias adaptadas, modificadas e testadas para avaliação da qualidade ecológica de reservatórios urbanos são ferramentas eficientes que poderão ser testadas em outros reservatórios brasileiros. Recomenda-se que os seguintes tópicos sejam avaliados futuramente:

1. Diagnosticar novas áreas com máximo potencial ecológico em outros reservatórios.
2. Definir um "Estado Ecológico" para as variáveis físicas, químicas e morfológicas a fim de obter-se um índice geral do "Estado Ecológico Total" para reservatórios.
3. Aplicar os índices de exergia em outros reservatórios utilizando comunidades bentônicas na região litorânea.

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